Current Trophic State and Potential Impacts of Coal Mine Development on Productivity of Middle Quinsam and Long Lakes

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CURRENT TROPHIC STATE AND POTENTIAL IMPACTS OF COAL MINE DEVELOPMENT ON PRODUCTIVITY OF MIDDLE QUINSAM AND LONG LAKES

by

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ABSTRACT

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Middle Quinsam and Long lakes are typical examples of the small oligotrophic lakes common to the coast of British Columbia. They share the common features of high rates of winter flushing, high water transparencies, very low concentrations of nitrogen and phosphorus, and low levels of pelagic productivity. Long Lake supports a 20 to 40% higher level of pelagic productivity than Middle Quinsam Lake, but a shallower morphometry and deeper light penetration suggest a higher level of benthic productivity in Middle Quinsam Lake. Long Lake experiences significant hypolimnetic oxygen depletion (0.8 mg $O_2 \cdot L^{-1} \cdot mo^{-1}$) during its growing season, but the current light climate of Middle Quinsam Lake permits hypolimnetic photosynthesis to maintain high oxygen levels in the hypolimnion. Inorganic nitrogen is depleted to detection limits for most of the growing season in both lakes. Based on nutrient bioassays and a comparison of summer chlorophyll:phosphorus yields, Middle Quinsam and Long lakes are slightly and moderately nitrogen-limited, respectively, during their growing seasons.

Middle Quinsam Lake is expected to respond to high-nitrogen loadings from future coal mine development with a 20 to 50% increase in algal production. This small increase in productivity may be beneficial to its salmonid production if current levels of phosphorus and water clarity are maintained. A doubling of algal productivity in Long Lake is predicted in response to nitrogen-rich discharges from mine operation. This level of productivity is sufficient to put Long Lake at significant risk of serious hypolimnetic oxygen depletion with attendent impacts on salmonids and their supporting food webs.

RÉSUMÉ

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Les lacs Middle Quinsam et Long sont des exemples typiques des petits lacs oligotrophiques communs sur la côte de la Colombie-Britannique. Dans ces lacs, le taux de lessivage en hiver est élevé, l'eau est très transparente, les concentrations d'azote et de phosphore sont très basses et le niveau de productivité pélagique est faible. Ce niveau est de 20 à 40% plus élevé dans le lac Long que dans le lac Middle Quinsam, mais la moins grande profondeur du lac Middle Quinsam et le fait que la lumière y pénètre plus profondément donnent à penser que la productivité benthique y est plus forte. L'hypolimnion du lac Long connaît un appauvrissement important en oxygène (0.8 mg O_2/l/mois) pendant la période de prolifération des algues, mais les conditions actuelles de lumière dans le lac Middle Quinsam permettent à la photosynthèse de maintenir des concentrations élevées en oxygène dans l'hypolimnion. L'azote inorganique est réduite aux limites de détection pendant la majeure partie de la saison de croissance des algues dans les deux lacs. D'après des tests biologiques sur des substances nutritives et une comparaison entre la production de chlorophylle et de phosphore pendant l'été, les lacs Middle Quinsam et Long sont respectivement légèrement et modérement limités par l'azote pendant la saison de production des algues.

Le lac Middle Quinsam devrait réagir à des charges élevées d'azote par suite de l'exploitation future de mines de charbon en augmentant de 20 à 50% la production d'algues. Cette faible augmentation de productivité pourrait favoriser la production de salmonidés dans ce lac si les concentrations actuelles de phosphore et la clarté de l'eau ne changent pas. On prévoit que la productivité sera deux fois plus grande par suite des déversements de substances riches en azote dus à l'exploitation minière. Ce niveau de productivité est suffisant pour que l'hypolimnion du lac Long risque un appauvrissement important en oxygène avec les répercussions qui s'ensuivent pour les salmonidés et les chaînes alimentaires dont ils dépendent.

INTRODUCTION

The salmonid populations of the Quinsam River system are a valuable renewable economic and social resource. Juvenile salmonids utilize the lakes and streams of the Quinsam system as rearing grounds, and during their freshwater residence, salmonids are particularly susceptible to changes in water quality or foraging resources induced by inputs of nutrients, metals, organics, acidic drainage, suspended solids, etc., caused by human activity within the drainage basin. The drainage basin of the Quinsam River system contains economic deposits of coal, and an open-pit coal mine has been proposed for future development. Among the potential impacts of mine development on the aquatic resources are nitrogen-rich discharges of pit drainage from settling ponds to Middle Quinsam and Long lakes.

Small carefully controlled inputs of nitrogen or phosphorus or both to lakes can increase primary and secondary production and be beneficial to salmonids by stimulating their supporting food webs and increasing their growth and survival (Hyatt and Stockner 1985). Conversely, uncontrolled inputs of nutrients can create excessive production with the attendant changes in supporting food web structure causing deleterious effects on both juvenile and adult salmonids (Stockner and Northcote 1974).

After formal public hearings on potential environmental impacts of coal mine development in the Quinsam system (Ministry of Environment 1984), an investigation of the two lakes to be affected by nutrient discharges was initiated by the Department of Fisheries and Oceans (DFO). This report presents the results of a limnological study of Middle Quinsam and Long lakes conducted by the Fisheries Research Branch from May to October 1984. The report specifically addresses two questions:

1. What is the current state of water quality in Middle Quinsam and Long lakes?

2. What are the potential impacts of nutrient-rich discharges from the proposed coal mine on water quality and aquatic biota of the lakes?

STUDY AREA

Middle Quinsam and Long lakes are located approximately 20 km southwest of the city of Campbell River on Vancouver Island, British Columbia (Fig. 1). The lakes and their drainage basins are in the Coastal Douglas-fir biogeoclimatic zone (Krajina 1973), and receive annual precipitation of 100-150 cm, largely concentrated in the winter months between October and March. Discrete drainage basins of Middle Quinsam and Long lakes are part of the larger Quinsam River drainage basin which was extensively logged in the 1950's. Middle Quinsam Lake receives the outflow from Long Lake as well as the runoff from the lakes and streams of the upper Quinsam River drainage basin. The latter inflow is subject to flow regulation for hydroelectric generation. The seasonal pattern of water circulation in the lakes is generally dimictic but the occurrence and duration of winter ice-cover can vary greatly from year to year.

MATERIALS AND METHODS

FIELD METHODS

Bathymetric maps of each lake were constructed from echo-sounding transects collected on October 23, 1984 in Middle Quinsam Lake and on October 25, 1984 in Long Lake. A Furuno FM-22A echo-sounder was used and morphometric data were calculated after Hutchinson (1957). Lake surface areas were determined from 1:50,000 topographic maps. Stream flow data collected at the outlet streams of each lake by staff of the Quinsam River Hatchery and Norecol Consultants Ltd., Vancouver, were used to calculate water-residence times and seasonal flushing for the October 1983 to October 1984 water-year. Flushing was assumed to be confined to the epilimnia of both lakes during thermal stratification from June through September.

The lakes were sampled every two weeks, from June 5, 1984 in Middle Quinsam Lake and June 26, 1984 in Long Lake, to October 16, 1984 and outlet streams were sampled monthly. Some limited sampling for winter (February 27) and early spring (April 17) nutrient levels in 1984 was also conducted in the lakes. An inflatable boat or float-equipped aircraft was used for sampling two stations in each lake (Fig. 1 and 2). Station 2 in Middle Quinsam Lake was not regularly sampled for vertical light and temperature profiles or for zooplankton densities.

Temperature profiles to maximum depth at station were obtained with a Montedoro-Whitney temperature probe (Model TC-5C). An equation of state (Chen and Millero 1977) was used to convert water temperature to densities for calculation of buoyancy frequencies (s^{-1}) (Turner 1973). The depth of maximum buoyancy frequency was used to determine epilimnion depth. A Li-Cor light meter (Model 185A) equipped with a Li-Cor underwater quantum sensor (Model Li-192S) was used to measure photosynthetically-active radiation (400-700 nm) at depth from lake surface to near-bottom. Vertical light-extinction coefficients (K_e) were calculated, and the compensation depth estimated as 1% of surface light intensity. A standard 22-cm white Secchi disk was used for water transparency.

Norecol Consultants Ltd., Vancouver, measured vertical profiles of dissolved oxygen (mg $O_2 \cdot L^{-1}$) at 1-m intervals to bottom in the central basin of each lake from April 1983 to October 1984. These data were used to assess oxygen levels in the hypolimnion of each lake and to calculate a volumetric hypolimnetic oxygen depletion (VHOD) rate for Long Lake. Oxygen concentrations from each depth were weighted for the volume of each stratum as calculated from bathymetry data. In Long Lake, only oxygen measurements below the compensation depth (10 m) were used to estimate a VHOD rate, to reduce direct contributions by photosynthesis and oxygen evolution in the upper hypolimnion.

Discrete water samples for chemical or biological measurements were collected with an opaque 6-L Van Dorn bottle. Samples were collected from depths of 1, 3 and 5 meters at each station with an additional sample from 8- to 12-meters depth depending on station depth. Screw-capped culture tubes were filled with 1% sulphuric acid prior to field use, autoclaved, rinsed thoroughly in deionized, distilled water (DDW) and dried. Each tube was rinsed and filled (25 ml) with sample water in the field, closed with cleaned aluminum foil, capped and stored at 4°C until analyzed for total phosphorus. Water samples for other nutrients, particulates and chlorophyll were collected in 1-L or 2-L

polyethylene bottles, kept cool and in the dark during transport, and processed in the field laboratory within 2 to 4 h.

LABORATORY METHODS

Water samples for dissolved nutrients were filtered through 47-mm diameter Whatman GF/F filters which had been ashed (460°C for 4 h) and washed (500 mL of DDW). Each filter was placed in a 47-mm Swinnex filter holder (Millipore Corp.), rinsed with an additional 500 mL of DDW and then rinsed with 50 to 75 mL of sample water. A precleaned (acid washed, DDW rinsed) glass bottle was rinsed and filled with 100 mL of filtered sample, covered with cleaned aluminum foil, capped and stored at 4°C in the dark for analyses of nitrate, ammonium, and total dissolved nitrogen. An additional 100 mL of sample was filtered into a precleaned and rinsed polyethylene bottle and stored at 4°C in the dark for analyses of soluble reactive silicate and total dissolved solids.

A 2-L sample was filtered onto an ashed and washed 47-mm diameter Whatman GF/F filter. Each filter was stored in a cleaned scintillation vial for analysis of particulate phosphorus. One liter of sample water was filtered onto an ashed 47-mm Whatman GF/F filter, folded in half in ashed aluminum dishes, and dried overnight in a dessicator. Filters were stored frozen for analyses of particulate carbon and nitrogen. A 500-mL sample was filtered under subdued light onto a 47-mm diameter 0.8-µm Millipore AA filter, and a few drops of $MgCO_3$ suspension were added. The filter was folded in half in an aluminum dish, dried overnight in a dessicator and stored frozen for analysis of chlorophyll.

All chemical analyses followed the methods described by Stephens and Brandstaetter (1983). Total phosphorus was measured after in-tube persulphate oxidation as the blue phospho-molybdenum complex (Traversy 1971). Nitrate was analyzed after cadmium-copper reduction as the azo dye (Brewer and Riley 1965) and ammonium as indophenol blue (Stainton et al. 1977). Total dissolved nitrogen was measured as ammonium after ultraviolet oxidation and soluble reactive silicate was analyzed by the acid molybdate reaction (Stainton et al. 1977).

Particulate phosphorus was measured after in-vial ignition (550°C) by a manual molybdenum blue method (Stainton et al. 1977). Particulate carbon and nitrogen were determined with a Perkin Elmer Model 240 Elemental Analyzer (Stainton et al. 1977). Fluorometry of 90% acetone extracts (Strickland and Parsons 1977) was used to measure chlorophyll without a phaeopigment subtraction.

Sterile test tubes containing 2-3 mL of 95% ethanol were rinsed and filled with sample water in the field for enumeration of total bacteria by the acridine orange direct count technique (Hobbie et al. 1977; MacIsaac et al. 1981). In the field laboratory, a polycarbonate filter (25-mm diameter, 0.2-µm pore size, Nuclepore Corp.), previously stained in irgalan black solution (2 $g \cdot L^{-1}$ in 2% acetic acid), was rinsed with 0.2-µm filtered deionized distilled water (FDDW) and placed on a wetted 40-µm mesh nylon screen (Nitex) in a 25-mm filter holder (Millipore Corp.). Five milliliters of sample water was added to the filter column and filtered at a vacuum not exceeding 20 cm Hg. Filters were removed when just dry, air-dried at room temperature in petri dishes lined with absorbent filter paper, and stored for enumeration at the main laboratory in West Vancouver. After 1 to 4 weeks of storage, each filter was placed on a wetted 40-µm mesh nylon screen in a filter holder and stained for 3 min. with 1

mL of 0.2-µm filtered acridine orange solution (0.05 $g \cdot L^{-1}$ in FDDW). The dye solution was drawn through (vacuum <20 cm Hg) and the filter was rinsed with 5 mL of FDDW. The moist filter was placed on a glass slide, a drop of fresh Cargille Type A immersion oil was placed on top, a coverslip was added and the bacteria were enumerated by epifluorescence microscopy at 1250 X magnification under oil immersion.

A Zeiss Model KLSM microscope equipped with a IV/FL epifluorescence condenser, a 50-W HBO mercury lamp, and exchangeable filter housings was used for all epifluorescence microscopy. Bacteria stained with acridine orange were enumerated under epifluorescence using a 450 to 490 nm band-pass exciter filter, a 510 nm beam-splitter mirror, and a 520 nm long-wave pass barrier filter. Random fields were counted on each filter until 300 bacteria or 10 fields were enumerated, and the counts were converted to number$\cdot mL^{-1}$. Occasional blanks were used to check for significant bacteria background counts in the dye solution and rinse water.

Opaque 125-mL polyethylene bottles were rinsed and filled with sample water in the field for enumeration of total phytoplankton. Phototrophic picoplankton (cyanobacteria and eukaryotic algae less than 3 µm) were counted with epifluorescence microscopy by filtering 15 mL of sample water through stained polycarbonate filters in the field laboratory as described for the collection of bacteria samples. Care was always taken to minimize exposure of the sample to light during sampling and laboratory processing. The filters were air-dried and stored in the dark at room temperature in opaque petri dishes. After 1 to 4 weeks of storage in the dark, the filters were prepared for enumeration under subdued light. Each filter was placed on a wetted 40-µm mesh nylon screen in a filter holder, 1 to 2 mL of FDDW was added to the filter column, and the cells on the filter were rehydrated for 3 to 5 min. The water was drawn through (<20 cm Hg) and the moist filter was placed on a glass slide with a drop of immersion oil and a coverslip.

The Zeiss epifluorescence microscope was equipped with a 397 nm longwave-pass exciter filter and a 560 nm shortwave-pass exciter filter, a 580 nm beam-splitter mirror, and a 590 nm longwave-pass barrier filter. Filters were examined at 1250 X magnification under oil immersion, and random fields were counted to a minimum of 200 cells or 30 fields per sample. Phototrophic picoplankton were identified as cyanobacteria or eukaryotic algae, assigned to general categories based on morphological characteristics and fluorescence color, and scored into size categories.

The remaining sample water in the opaque polyethylene bottles was fixed with 1 mL of Lugol's acid solution for enumeration of nano (3 to 20 µm) and net (>20 µm) phytoplankton. Each sample was mixed and a subsample settled overnight in a settling chamber of 7-, 12-, or 27-mL volume. One transect at 187.5 X and one at 750 X magnification were counted using a Wild M40 inverted microscope equipped with phase contrast optics. Cells were identified to genus or species and assigned to one or more size classes. Counts were converted to numbers and cell volumes, and carbon biomass was calculated using formulas modified from Strathmann (1967).

Zooplankton were sampled by vertical hauls with a 100-um mesh dark SCOR[1] net (0.25-m^2 mouth area) towed at approximately 0.5 $m \cdot s^{-1}$ from near-bottom to the surface. Zooplankton from each net haul were preserved in a 4% formalin-sucrose solution buffered with borax (Haney and Hall 1973). Filtration efficiency of the net was assumed to be 100%. Each sample was halved

using a Folsom plankton splitter. One half was filtered onto an ashed and weighed Whatman GFC filter then dried to a constant weight at 90°C for 24 h. The filter was weighed, ashed at 460°C for 4 h, and weighed again. Zooplankton biomass was calculated as mg ash-free dry weight·m^{-3} (AFDW).

The other half of the zooplankton sample was used for zooplankton enumeration and sizing using a microcomputer-based caliper system (Sprules et al. 1981). Sample volume was increased to 300 or 900 mL, mixed then subsampled with a wide-mouth pipet. Volume of the subsample varied (3 to 9 mL) depending on the density of zooplankton in the sample so that at least 300 individuals were counted and sized. The subsample was returned to the original split sample and half or all of the split sample was then scanned for counting and sizing rare and/or large (>1 mm) organisms. Zooplankton were measured and usually identified at least to genus. Enumeration data were stored on floppy disks and processed by an Apple II microcomputer. Biomass based on length-weight regressions (Edmondson and Winberg 1971) and total numbers were calculated for each taxa in specified size categories as numbers·m^{-3} and mg wet weight·m^{-3}.

BIOASSAY METHODS

Water for nutrient bioassays was collected on July 11 and September 18 from Middle Quinsam Lake (station 1) and on August 21 from Long Lake (station 1). A clean 20-L carboy was rinsed and filled with water from 1-m depth, stored cool and in the dark, and immediately transported to the main laboratory at West Vancouver. Bioassays were started within 20 h of initial sample collection. Screw-capped polycarbonate erlenmeyer flasks (300 mL) were cleaned in 1% Liquinox detergent, 1% hydrochloric acid, and rinsed in 0.2-µm filtered deionized distilled water (FDDW). Each of 12 flasks was rinsed and filled with a 250-mL aliquot of sample water from the mixed carboy and acclimated for 24 h in a Percival Incubator (Illuminated Model I35LLVL) prior to nutrient additions. Flasks were incubated at the temperature of the sample water at initial collection (16 to 20°C) and illuminated with cool-white fluorescent bulbs at 80 µEinst·m^{-2}·s^{-1} on a 16:8 h light:dark cycle.

Each bioassay consisted of two replicates each of a control (no additions) and five nutrient treatments:

1. chelated trace elements only (enriched with 0.8 µmol Fe, 0.12 µmol Mn, 0.8 µMol B, 0.024 µmol Zn, 0.024 µmol Mo, 0.016 µmol Co, 0.004 µmol Cu, chelated with 2.0 µmol EDTA).
2. vitamins only (enriched with 0.4 µmol Thiamin, 0.004 µmol Biotin and 0.001 µmol Cyanocobalamin).
3. nitrogen only (enriched with 10 µmol NO_3^- and 2 µmol NH_4^+).
4. phosphorus only (enriched with 0.3 µmol PO_4^{3-}).
5. nitrogen and phosphorus (enriched with 10 µmol NO_3^-, 2 µmol NH_4^+ and 0.3 µmol PO_4^{3-}).

Nutrient stock solutions were made with analytical-grade chemicals dissolved in FDDW, autoclaved and stored at 4°C or frozen (vitamins). Treatment flasks were inoculated with 0.2 to 0.5 mL of the appropriate stock solution(s), capped and mixed.

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[1] Scientific Committee on Ocean Research.

Flasks were randomly arranged on two shelves in the incubator and rotated among rows and shelves every two days. Algal growth was monitored on the day of acclimation and every two days thereafter by in vivo chlorophyll fluorescence. Each flask was mixed and the chlorophyll fluorescence of a 4-mL aliquot was measured in a Turner Model 111 fluorometer. The fluorometer was zeroed against a FDDW blank. After 30 to 40 days incubation, each flask was mixed and harvested for final-biomass yield. Samples of 5, 10 and 20 mL were taken for bacteria, phototrophic picoplankton and phytoplankton, respectively, and preserved and enumerated as described in the Field Methods. Culture tubes were also rinsed and filled for total phosphorus determinations as previously described. Particulate carbon was collected on ashed and washed 47-mm diameter Whatman GF/F filters. Two filters were placed, one on top of the other, in a 47-mm polycarbonate filter holder (Nalgene) and wetted with FDDW. A 100-mL aliquot was filtered through (<20 cm Hg) and each filter was removed under reduced vacuum, folded in an ashed aluminum dish, dessicated under vacuum over silica gel overnight, and stored frozen. Particulate carbon values for the bottom filter were subtracted from those of the top filter to correct for "dissolved" organic carbon retained by glass-fiber filters (Menzel and Dunstan 1973).

RESULTS

MORPHOMETRY

Middle Quinsam Lake is approximately 2.5 km long and 500 m wide with a mean depth of 5 m and a maximum depth of 15 m (Table 1, Fig. 2). Approximately 60% of total lake surface area overlies water <5-m depth and there are extensive shallows <3-m depth at the west end of the lake. Long Lake is smaller and deeper than Middle Quinsam Lake, with a length of 1.3 km, width of 150 m, mean depth of 7 m and a maximum depth of 18 m (Table 1, Fig. 3). The littoral zone is less extensive in Long Lake with approximately 35% of lake surface area overlying water <5-m depth.

HYDROLOGY

Drainage basin areas in the vicinity of Campbell River receive an average annual precipitation (20-y normal) of approximately 150 cm of which 100 cm is available for runoff after evaporation and evapotranspiration (Fig. 4). Only about 10% of this runoff occurs during the May to October growing season. However the seasonal storm patterns of coastal British Columbia are unpredictable and there can be large between-year variations in annual and seasonal runoffs.

Middle Quinsam Lake receives the outflow of Long Lake at its east end near its outflow and receives regulated minimum inflows from the Quinsam River at its west end. Theoretical water residence time calculated on an annual basis is 0.05 y (Table 1) but there are large seasonal variations in percent water renewal (Fig. 5). Flushing is largely confined to the epilimnion from June to September with monthly percent water renewals in the epilimnion of 40 to 150%. Winter flushing (November to March) is thorough, with percent water renewals

ranging from 70% up to 700% in some months.

Long Lake does not receive the benefits of regulated minimum flows during its growing season, and monthly percent water renewals of <5% are common for the epilimnion during summer months (Fig. 5). Winter flushing is also thorough in Long Lake with monthly percent water renewals ranging from 90 to 400%, accounting for its annual water residence time of 0.07 y.

THERMAL STRATIFICATION

Middle Quinsam Lake was weakly stratified in early June with a broad thermocline at about 5 m and with epilimnion temperatures of 14 to 15°C and relatively warm hypolimnion temperatures of 12 to 13°C (Fig. 6). Warming of the surface waters increased epilimnion temperatures to 21 to 23°C by late July and strengthened the thermocline at a depth of about 6 m. Subsequent wind mixing pushed the thermocline to 9.5 m by early September with cooler epilimnion temperatures of about 17°C and warmer hypolimnion temperatures of 13 to 15°C. By late September near-isothermal conditions had developed with complete mixing and cooling of the lake to 11°C by mid-October.

Long Lake had a strong thermocline at 4 m when first sampled in June and strong thermal stratification was maintained until near isothermal conditions (8-10°C) developed in mid-October (Fig. 6). The depth of the epilimnion remained at 4 to 6 m until early September with mean temperatures of 16 to 20°C. Wind mixing in the fall increased the epilimnion depth to 7.5 m with mean temperatures of 12 to 14°C. Hypolimnion temperatures were relatively constant throughout the sampling season at 7 to 9°C.

LIGHT

Middle Quinsam Lake has a mean euphotic zone (compensation) depth of 17 m that exceeds its maximum lake depth of 15 m (Table 2, Fig. 7). The euphotic zone ranges from 8 to 15 m in the fall and early summer to a maximum of 20 to 35 m (extrapolated) in mid-summer. Secchi depths ranged from 8 to 10 m in summer to 6 to 8 m in the fall, but showed little correspondence with seasonal changes in the measured extinction of photosynthetically active radiation.

Euphotic zone depth of Long Lake averaged 9 m compared to a maximum lake depth of 18 m (Table 2, Fig. 8). The depth of the euphotic zone was relatively constant during the sampling season, ranging from 8 to 13 m with Secchi depths of 6 to 7 m.

OXYGEN

Significant depletion of hypolimnetic oxygen during the growing season occurred only in Long Lake. Volume-weighted oxygen concentrations below the euphotic zone (tropholytic zone) ranged from 9-10 mg $O_2 \cdot L^{-1}$ in April-May to a low of 5-6 mg $O_2 \cdot L^{-1}$ in September-October (Fig. 9). A volumetric hypolimnetic oxygen depletion (VHOD) rate of 0.8 mg $O_2 \cdot L^{-1} \cdot mo^{-1}$ was estimated from the linear regression of average oxygen concentration versus time. Oxygen measurements near the sediments (>10 m) reached levels <4 mg $O_2 \cdot L^{-1}$ near the end of the growing season. Low oxygen concentrations (<6 mg $O_2 \cdot L^{-1}$) in Middle Quinsam Lake were only recorded at depths near the sediments (>11 m) in September at the end of the growing season.

NUTRIENTS

Total phosphorus levels averaged 2µg $P \cdot L^{-1}$ during the winter and early spring in Middle Quinsam Lake (Table 3). Early summer concentrations of total phosphorus were 2 to 3 µg $P \cdot L^{-1}$ declining to <1 to 2 µg $P \cdot L^{-1}$ in August and September (Table 2, Fig. 10). Ammonium concentrations during the growing season were always at or below the detection limit of 4 µg $N \cdot L^{-1}$. Winter nitrate levels of 16 µg $N \cdot L^{-1}$ were significantly depleted to 1 µg $N \cdot L^{-1}$ by April (Table 3). Nitrate levels remained at or near the detection limit of 1 µg $N \cdot L^{-1}$ for most of the growing season, although levels as high as 3 µg $N \cdot L^{-1}$ were occasionally recorded during August or September. With the breakdown of stratification in mid-October, both nitrate and ammonium levels increased to about 8 µg $N \cdot L^{-1}$.

Total phosphorus levels in Long Lake ranged from 2 to 4 µg $P \cdot L^{-1}$ in winter through early summer to 1 to 3 µg $P \cdot L^{-1}$ during August and September, and average concentrations of total phosphorus were approximately 30% higher than in Middle Quinsam Lake (Tables 2 & 3, Fig. 11). Winter nitrate levels of 19 µg $N \cdot L^{-1}$ were depleted to 5 µg $N \cdot L^{-1}$ in April. Epilimnetic nitrate and ammonium levels were always near or at the detection limit when Long Lake was thermally stratified. Seasonally averaged concentrations of nitrate and ammonium were near the detection limits and similar to averaged levels in Middle Quinsam Lake.

Levels of soluble reactive silicon during the growing season averaged 1500 µg $Si \cdot L^{-1}$ in Middle Quinsam Lake and 2400 µg $Si \cdot L^{-1}$ in Long Lake (Table 2). Seasonal depletion of silicon was slight in both lakes. Total dissolved solids concentrations of 30 to 32 $mg \cdot L^{-1}$ in the lakes are consistent with other lakes and streams of the central east coast of Vancouver Island (Stockner et al. 1980).

PARTICULATES

Concentrations of phosphorus, nitrogen and carbon in the seston of Middle Quinsam and Long lakes exhibited relatively little seasonal variation during the sampling season (Fig. 12 and 13). Particulate phosphorus levels ranged from 1 to 2 µg $P \cdot L^{-1}$ in both lakes while particulate nitrogen ranged from 20 to 30 µg $N \cdot L^{-1}$ in Middle Quinsam Lake and 25 to 35 µg $N \cdot L^{-1}$ in Long Lake. Particulate carbon levels were 150 to 250 µg $C \cdot L^{-1}$ in Middle Quinsam and 200 to 300 µg $C \cdot L^{-1}$ in Long Lake, with slightly higher values in the fall in both lakes. Concentrations of particulates averaged 10 to 40% higher in Long than in Middle Quinsam Lake (Table 2).

PLANKTON BIOMASS

Bacterioplankton, phytoplankton and zooplankton biomass were low in Middle Quinsam and Long lakes and indicative of the current oligotrophic state of the lakes (Table 2). Plankton standing stocks averaged 15 to 35% higher in Long Lake than in Middle Quinsam Lake, suggesting a higher level of pelagic productivity in Long Lake.

Marked seasonal patterns in plankton biomass were not apparent in either lake (Fig. 14 and 15). Bacterioplankton numbers ranged from 1 to 1.5 $\cdot 10^6 \cdot mL^{-1}$ in Middle Quinsam Lake with highest densities generally noted in mid-summer, while in Long Lake bacteria levels were higher and ranged from 1 to $1.9 \cdot 10^6 \cdot mL^{-1}$, with highest numbers in late summer and fall. Levels of

chlorophyll and zooplankton biomass were more variable than bacteria densities during the growing season in both lakes. Chlorophyll ranged from 0.6 to 1.4 $\mu g \cdot L^{-1}$ with 30 to 120 mg AFDW$\cdot m^{-3}$ of zooplankton biomass in Middle Quinsam Lake. In Long Lake a maximum of 2 $\mu g \cdot L^{-1}$ chlorophyll was reached in August, and zooplankton biomass declined from a spring maximum of 160 to a fall minimum of 40 mg AFDW$\cdot m^{-3}$.

PHYTOPLANKTON

Phytoplankton biomass (as carbon) ranged from 60 to 100 mgC$\cdot m^{-3}$ in Middle Quinsam Lake (Fig. 16), with chrysophytes, principally the flagellated Chromulina sp. and Mallomonas sp., dominating the phytoplankton community for much of the year. Cyanobacteria were the next largest contributors to carbon biomass during the summer, with the picoplankton Synechococcus sp. and small, colonial gelatinous taxa particularly abundant. Nitrogen-fixing cyanobacteria, notably Anabaenopsis sp., Aphanizomenon sp. and Anabaena sp. were frequently encountered during the summer months. Diatoms were not a major contributor to phytoplankton biomass in Middle Quinsam Lake.

Phytoplankton biomass in Long Lake was higher than in Middle Quinsam Lake, ranging from 100 to 150 mgC$\cdot m^{-3}$, with chrysophytes and cyanobacteria dominant (Fig. 16). The chrysophyte and cyanobacteria taxa noted in Middle Quinsam Lake were very similar to those observed in Long Lake, but nitrogen-fixing cyanobacteria were more prevalent in Long Lake. Diatoms only made a significant contribution to phytoplankton biomass in July concommitant with a small bloom of Asterionella formosa.

Seasonal averages of phytoplankton carbon biomass in Middle Quinsam and Long lakes were 85 and 125 mgC$\cdot m^{-3}$, respectively. Compared to average particulate carbon concentrations determined by chemical analyses for the seston in each lake (Table 2), the microscopic carbon estimates suggest that phytoplankton biomass contributes only 40 to 55% of the total particulate carbon collected by glass-fiber filters in the lakes.

ZOOPLANKTON

Rotifers were numerically abundant in Middle Quinsam Lake reaching a maximum of 19000$\cdot m^{-3}$ in the fall (Fig. 17), and dominant species were Gastropus sp., Conochilus sp., Kellicottia sp., and Keratella sp.. Cladocera densities ranged from 1100 to 7500$\cdot m^{-3}$, with Daphnia rosea dominant. Copepods reached a summer maximum of 11000$\cdot m^{-3}$ consisting almost exclusively of Diaptomus oregonensis. Of particular importance to size-selective foraging salmonids was the abundance of large-bodied taxa such as Chaoborus sp. and chironomidae larvae in Middle Quinsam Lake.

Rotifer densities in Long Lake reached 24000$\cdot m^{-3}$ with taxa similar to those found in Middle Quinsam Lake (Fig. 17). Cladocera densities ranged from 2200 to 8800$\cdot m^{-3}$, with Daphnia rosea and D. longiremis numerically dominant. Diaptomus oregonensis was the dominant copepod, with total numbers ranging from 1400 to 5000$\cdot m^{-3}$. As in Middle Quinsam Lake, Chaoborus sp., chironomidae larvae and other large taxa were frequently found in Long Lake samples.

NUTRIENT BIOASSAYS

Bioassays for nutrient limitation of phytoplankton in Middle Quinsam

Lake showed similar results in both July and September experiments (Fig. 18 and 20). There was little or no response shown to additions of trace elements or vitamins relative to the controls. Positive responses to nitrogen additions alone were apparent in both bioassays with the final yield of bacteria, chlorophyll fluorescence and particulate carbon averaging 20 to 50% higher than controls. Treatment responses to phosphorus additions alone were approximately 2 to 4 fold higher than controls. The strong responses to phosphorus additions in both bioassays reflected the presence and enhanced growth of nitrogen-fixing Anabaena sp. and Anabaenopsis sp.. Additions of both nitrogen and phosphorus gave the greatest response, yielding 5 to 10 times higher biomass than controls.

Phytoplankton responses to nutrient additions in Long Lake in August were similar but stronger than the treatment responses of Middle Quinsam Lake phytoplankton (Fig. 19 and 20). Additions of trace elements or vitamins did not stimulate phytoplankton growth relative to the controls. Nitrogen additions resulted in a doubling in phytoplankton biomass. The strong responses to phosphorus additions was by nitrogen-fixing Anabaena sp. and Anabaenopsis sp. similar to that observed in the Middle Quinsam Lake bioassays. Enrichment with both nitrogen and phosphorus gave the highest growth response, with a 5 to 10 fold increase over the controls.

DISCUSSION

THE CURRENT STATE OF THE LAKES

Middle Quinsam and Long lakes are typical coastal Vancouver Island lakes, sharing in common with all lakes in this biogeographic zone the following characteristics: (a) fast-flushing (short water renewal times), (b) clear water (high transparency with low light attenuation), (c) nutrient-poor conditions (low concentrations of dissolved ions) and (d) low ambient productivity (oligotrophic). Low chemical weathering and leaching from the shallow soils and granitic bedrock of the drainage basins produce extremely low concentrations of both phosphorus and nitrogen in input sources to each lake, and rapid complete flushing during winter ensures a low nutrient retention rate.

Low bacterioplankton, phytoplankton and zooplankton standing stocks in the lakes during the growing season reflect the oligotrophic condition and suggest that pelagic productivity is 20 to 40% higher in Long Lake than in Middle Quinsam Lake. However, differences in morphometry and light climate play an important role in the relative productivities of the two lakes. Middle Quinsam Lake is relatively shallow with significant light penetration to maximum depth, while Long Lake is deeper with an average compensation depth extending only a few meters beneath the thermocline. Thus, primary production by hypolimnetic phytoplankton and benthic algae and macrophytes makes a much greater contribution to lake productivity in Middle Quinsam Lake, and may compensate for the lower level of pelagic productivity. Significant photosynthesis in the hypolimnion is also likely responsible for observed higher oxygen levels of hypolimnetic water during the growing season in Middle Quinsam Lake. Deep light penetration and absorbtion by hypolimnetic water and sediments may also account for the relatively warm hypolimnion of Middle Quinsam Lake (Bachmann and Goldman 1965). In summary, the current light climate of Middle

Quinsam Lake has a major effect on the current physical and biological characteristics of Middle Quinsam Lake.

NUTRIENT LIMITATION

Large increases in nitrogen loadings to the Quinsam system are a certainty from the use of nitrogen-based explosives during mine operation (Ministry of Environment 1984), and significant increases in phosphorus loadings may be generated during initial mine development and operation. The potential impacts of these changes in nutrient loadings on the fisheries and aquatic resources of the Quinsam system depends on identification of the nutrient or nutrients that currently limit algal productivity. Our evaluation of nutrient limitation is based on 3 factors: (1) 1984 epilimnetic nitrogen and phosphorus concentrations, (2) algal nutrient bioassay results, and (3) a comparison of phytoplankton biomass (chlorophyll): phosphorus yields in Middle Quinsam and Long lakes with other Vancouver Island coastal lakes. A consideration of evidence from each follows:

1. From June through September inorganic nitrogen was depleted to values at or below the analytical detection limit (<1 µg NO_3-N·L^{-1}; <4 µg NH_4-N·L^{-1}) in both Middle Quinsam and Long lakes, indicating high nitrogen demands by the autotrophic communities.
2. Nutrient bioassay experiments suggest slight nitrogen limitation in Middle Quinsam Lake, while the stronger growth responses to nitrogen additions in Long Lake indicate moderate nitrogen limitation. However, it should be noted that with nitrogen additions, very low ambient phosphorus concentrations only allowed up to a doubling in growth (biomass yield) before it became the limiting nutrient.
3. The average yield of algal biomass (chlorophyll) per unit phosphorus of Middle Quinsam and Long lakes in 1984 was compared to the extensive database of the DFO Lake Enrichment Program on chlorophyll:phosphorus yields of oligotrophic and nutrient-enriched Vancouver Island lakes (Stockner and Shortreed 1985). When compared on the regression plot of average chlorophyll versus total phosphorus concentrations in these other Vancouver Island lakes, Middle Quinsam and Long lakes have chlorophyll:phosphorus ratios that are 50 and 70% of the expected average yields (Fig. 22). These data support the bioassay results and suggest that slight to moderate nitrogen limitation may be limiting the yield of algal biomass per unit of phosphorus in both lakes.

The growth-response and final-yield bioassays should be given the greater weight in our nutrient evaluation because they are direct experimental determinations of algal nutrient limitation. The bioassays indicate slight nitrogen-limitation in the epilimnion of Middle Quinsam Lake and moderate nitrogen-limitation in Long Lake with both final yields constrained by low phosphorus concentrations. This principle finding is supported by the observed duration of nitrate and ammonium exhaustion in both lakes and by the phytoplankton data which indicate the presence of heterocystic nitrogen-fixing cyanobacteria in both lakes. However, the bioassays also indicate the eventual importance of both nitrogen and phosphorus in limiting algal productivity and species composition in both Middle Quinsam and Long lakes.

POTENTIAL IMPACTS

Elevated nutrient loads from proposed mining activities may have detrimental impacts on the aquatic resources of the Quinsam system.

Uncontrolled increases in nutrients in both lakes and streams may exceed the capacity of the current communities to assimilate excessive nutrients and the anticipated additional production. The potential for detrimental changes in food web structure and diminished trophic efficiency will increase as nutrient loads rise by:

1. altering the species/size structures of the pelagic and benthic algal communities and the zoobenthic and zooplanktonic communities they support (Brooks 1969, Jonasson 1969, Crone 1981),
2. shifting the current relative contributions of both pelagic and benthic food-webs to fish productivity, and by
3. depleting oxygen levels in the sediments and hypolimnia of both lakes which may release sediment-bound phosphorus (internal nutrient loading), eliminate part of the benthic community (Jonasson 1969), and eliminate the thermal refuge for zooplankton and fish (Davis 1975).

These impacts on food-web structure generally herald declines in aesthetic and recreational water quality and changes in the quality and quantity of fish production.

An initial evaluation of impacts can be made by assuming that the only significant nutrient impact on the Quinsam lakes will be from elevated nitrogen loading. The initial impact of elevated nitrogen concentrations that may reach 2 to 4 $mg\text{-}N\cdot L^{-1}$ in surface waters (Ministry of Environment 1984) will be a shift from slight or moderate nitrogen-limited algal growth to severely phosphorus-deficient growth, with ambient phosphorus levels setting primary production limits. Based on the bioassay results and expected chlorophyll: phosphorus yields, increases in summer algal biomass will be in the order of 20 to 50% in Middle Quinsam Lake (1.1 to 1.4 $\mu g\cdot L^{-1}$ chlorophyll) and up to a doubling in Long Lake (up to 2.5 $\mu g\cdot L^{-1}$ chlorophyll). Shifts in phytoplankton species will likely include reductions in cyanobacteria populations and a probable increase in diatom densities. These predicted changes are very similar to those that occurred in Mohun Lake, another lake in the Campbell River vicinity that was subject to high nitrogen loadings (1.0-1.5 $mg\cdot L^{-1}$) from forest fertilization activities in 1980 (Perrin et al. 1984). Mohun Lake responded to nitrogen enrichment with an approximate doubling in chlorophyll concentrations.

The anticipated increases in algal productivity in Middle Quinsam Lake are relatively small and unlikely to significantly affect hypolimnion or surface sediment oxygen demand. Productivity increases of this magnitude may be beneficial to rearing salmonids by stimulating benthic and pelagic food production without reducing light penetration and oxygen evolution in the hypolimnion. However, increased nitrogen loads must not be accompanied by significant increases in phosphorus, organics, or suspended solid loadings with associated increases in turbidity. Increased concentrations of suspended solids or algal biomass would reduce light penetration, and the latter (algae) has the added complication of increasing organic sedimentation and oxygen demand in the hypolimnion. Significant reductions in light penetration (to a compensation depth <8 m) will also shift the current benthic/pelagic balance away from the benthic food webs. Calculation of a critical threshold level for phosphorus in the surface waters of Middle Quinsam Lake is therefore a complicated product of anticipated changes in light climate (dependent on turbidity and algal biomass) and increased hypolimnetic oxygen demand (dependent on autochthonous and allochthonous organic loading). A maximum response of a doubling of algal

productivity is anticipated in Long Lake if only nitrogen loading increases. A volumetric hypolimnetic oxygen depletion (VHOD) rate of 0.8 mg $O_2 \cdot L^{-1} \cdot mo^{-1}$ was calculated for the current trophic state of Long Lake (Fig. 9), sufficient to deplete hypolimnion oxygen levels to 5-6 mg $O_2 \cdot L^{-1}$ by September-October. Assuming that VHOD rates are proportional to the productivity of the lake (Janus and Vollenweider 1982), a doubling of the productivity of Long Lake would increase the VHOD rate to 1.6 mg $O_2 \cdot L^{-1} \cdot mo^{-1}$. Assuming an initial spring (April) oxygen concentration of10 mg $O_2 \cdot L^{-1}$ (Fig. 9), this VHOD rate is sufficient to deplete oxygen levels in the hypolimnion to below 2 mg $O_2 \cdot L^{-1}$ by September. A doubling of algal biomass in the surface waters would also reduce light penetration to the upper hypolimnion, eliminate hypolimnetic photosynthesis as a current internal source of oxygen and increase the VHOD rate. Thus, additional loadings of phosphorus, suspended solids or organics are not required to put Long Lake at risk of serious hypolimnetic oxygen depletion. Any increments in increased phosphorus loadings will shift the probability of hypolimnion and sediment oxygen deficits from a significant risk to a certainty. Aggravated by the possible release of sediment-bound phosphorus, anticipated changes in benthic and pelagic food web structure and elimination of a thermal refuge for zooplankton and fish are collectively likely to have a detrimental impact on the growth and survival of the salmonids which utilize Long Lake.

CONCLUSION

In their current state, Middle Quinsam and Long lakes experience slight and moderate nitrogen-limitation respectively during the growing season. Increased nitrogen levels during mine operation will stimulate algal production by 20 to 50% in Middle Quinsam Lake and up to a doubling in Long Lake. Further productivity increases will be dependent on increases in phosphorus loadings. These relatively small increases in productivity levels in Middle Quinsam Lake may be beneficial to the growth and survival of salmonids, contingent on negligible changes in lake-water turbidity, phosphorus loading and organic loading during the growing season. A doubling in algal productivity is anticipated in Long Lake if only nitrogen loading increases. Anticipated increases in phosphorus loadings are not required to put Long Lake at significant risk of serious hypolimnetic oxygen depletion. Any increments in increased phosphorus loadings will shift the probability of serious oxygen deficits from a significant risk to a certainty. The accompanying changes in food web structure and elimination of the hypolimnetic thermal refuge in Long Lake are expected to have serious detrimental impacts on the growth and survival of salmonids.

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Table 1. Geographic, morphometric and hydrologic characteristics of the study lakes.

	Middle Quinsam Lake	Long Lake
Latitude	125° 29'	125° 28'
Longitude	49° 55'	49° 55'
Surface area (ha)[a]	78.4	13.7
Volume ($10^6 m^3$)[a]	3.64	0.93
Maximum depth (m)[a]	15.2	17.9
Mean depth (m)[a]	4.6	6.8
Water Residence Time (y)[b]	0.05[c]	0.07

[a] Calculated from bathymetric data collected on October 23 to 25, 1984.

[b] Calculated from outflow data for October 1983 to October 1984 collected by DFO Quinsam River Hatchery and Norecol Consultants Ltd., Vancouver.

[c] Regulated flows.

Table 2. Seasonal means and ranges (June-October) of physical, chemical and biological variables, measured during 1984.

Parameters	Middle Quinsam Lake		Long Lake	
Mean (Range)	Station 1	Station 2	Station 1	Station 2
Surface temperature (°C)	17.4 (11.0-23.0)	-	17.2 (10.2-22.0)	17.1 (10.2-21.8)
Epilimnion temperature[b] (°C)	16.8 (10.6-21.9)	-	15.8 (9.8-18.4)	16.1 (9.8-19.8)
Epilimnion depth[a] (m)	8.0 (5.4-iso.)	-	6.0 (3.8-9.4)	5.8 (3.5-9.6)
Compensation depth (m)	16.9 (8.0-34.3)	-	9.5 (7.3-12.9)	9.2 (7.0-12.5)
Extinction coefficient (k_e)	0.31 (0.13-0.55)	-	0.47 (0.37-0.59)	0.49 (0.39-0.60)
Secchi depth (m)	8.3 (6.0-9.5)	-	6.4 (6.0-7.0)	6.1 (5.0-7.0)
Total Dissolved Solids[b] ($mg \cdot L^{-1}$)	32.0 (30.7-34.0)	32.6 (27.6-35.2)	29.9 (26.8-33.7)	30.6 (28.8-34.9)
Soluble Reactive Silicon[b] ($\mu g\ Si \cdot L^{-1}$)	1478 (1318-1646)	1457 (1273-1624)	2442 (2294-2572)	2467 (2280-2712)
Total Dissolved Nitrogen[b] ($\mu g\ N \cdot L^{-1}$)	198 (140-297)	221 (157-409)	245 (193-315)	247 (183-472)
Nitrate-Nitrogen[bc] ($\mu g\ N \cdot L^{-1}$)	1.9 (<1.0-7.2)	2.3 (<1.0-8.5)	1.8 (<1.0-6.8)	1.8 (<1.0-6.8)
Ammonium-Nitrogen[bc] ($\mu g\ N \cdot L^{-1}$)	4.4 (<4.0-7.8)	4.5 (<4.0-7.8)	4.3 (<4.0-6.0)	5.1 (<4.0-14.0)
Total Phosphorus[bc] ($\mu g\ P \cdot L^{-1}$)	1.8 (<1.0-2.5)	2.1 (<1.0-4.3)	2.5 (<1.3-4.2)	2.7 (<1.3-4.0)
Particulate Phosphorus[b] ($\mu g\ P \cdot L^{-1}$)	1.3 (0.8-1.6)	1.4 (0.9-2.0)	1.8 (0.7-3.2)	1.9 (1.6-2.8)
Particulate Nitrogen[b] ($\mu g\ N \cdot L^{-1}$)	25.1 (18.5-35.5)	25.5 (22.0-33.1)	28.7 (25.2-33.4)	27.7 (24.0-31.9)

Table 2. Continued

Parameters	Middle Quinsam Lake		Long Lake	
Mean (Range)	Station 1	Station 2	Station 1	Station 2
Particulate Carbon[b] ($\mu g\ C \cdot L^{-1}$)	202 (152-266)	211 (186-247)	231 (188-258)	240 (202-281)
Total Bacteria[b] ($10^6 \cdot mL^{-1}$)	1.17 (0.90-1.48)	1.28 (0.88-1.65)	1.46 (0.92-2.16)	1.39 (0.93-1.66)
Chlorophyll[b] ($\mu g \cdot L^{-1}$)	0.94 (0.64-1.45)	0.85 (0.56-1.31)	1.19 (0.78-1.94)	1.24 (0.75-2.05)
Zooplankton AFDW ($mg \cdot m^{-3}$)	62.6 (29.5-118.6)	-	69.9 (42.5-106.4)	86.9 (33.2-235.9)

[a] Epilimnion depth calculated as the depth of maximum stability based on buoyancy frequencies.

[b] Epilimnetic means calculated using values above depth of maximum stability or all depths when water column was not strongly stratified.

[c] Values below the analytical detection limits were assumed equal to the detection limit when means were calculated. Calculated means are therefore overestimates.

Table 3. Winter and early spring (1984) nutrient levels in Middle Quinsam and Long Lakes. Reported as means of samples from 3 or 4 depths at a single mid-lake station or as a single outlet-stream sample.

Parameter	Middle Quinsam Lake		Long Lake	
	February 27	April 17	February 27	April 17[a]
Total phosphorus ($\mu g\ P \cdot L^{-1}$)	1.7	2.3	2.0	4
Nitrate ($\mu g\ N \cdot L^{-1}$)	16	1	19	5

[a] Single sample for outlet-stream.

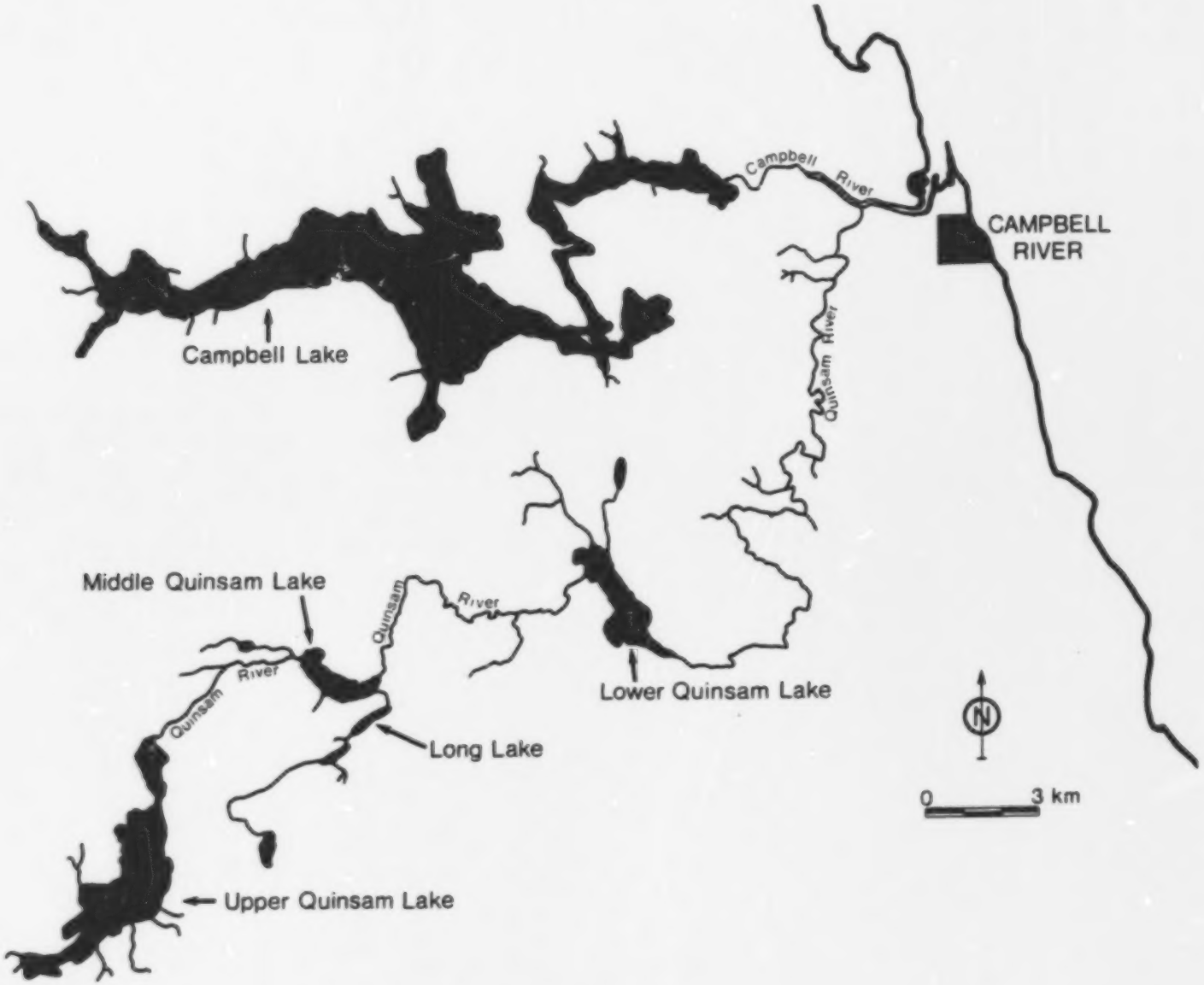

Fig. 1. Map of Quinsam River system and study lakes.

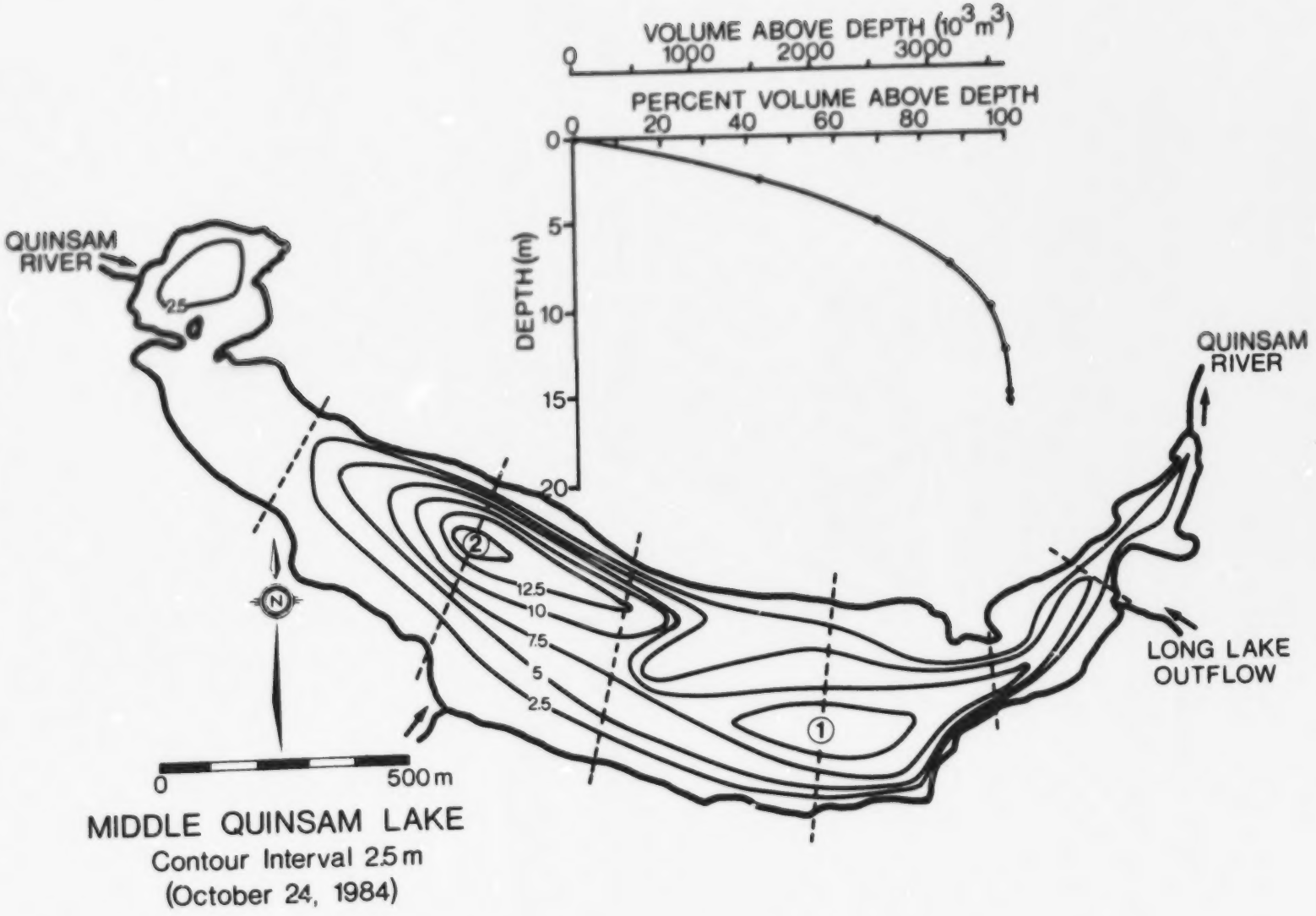

Fig. 2. Bathymetric map of Middle Quinsam Lake and depth-volume curve.

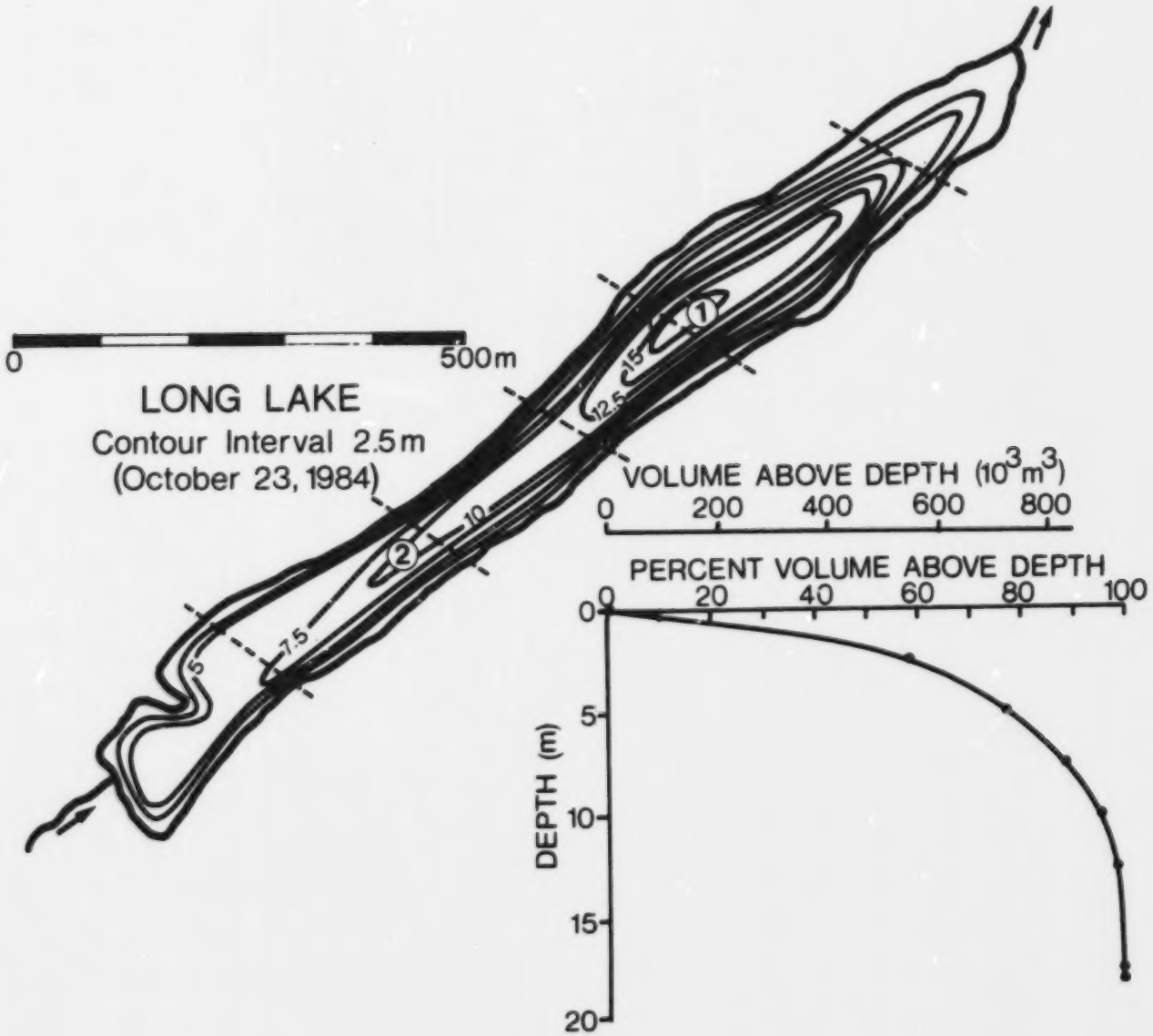

Fig. 3. Bathymetric map of Long Lake and depth-volume curve.

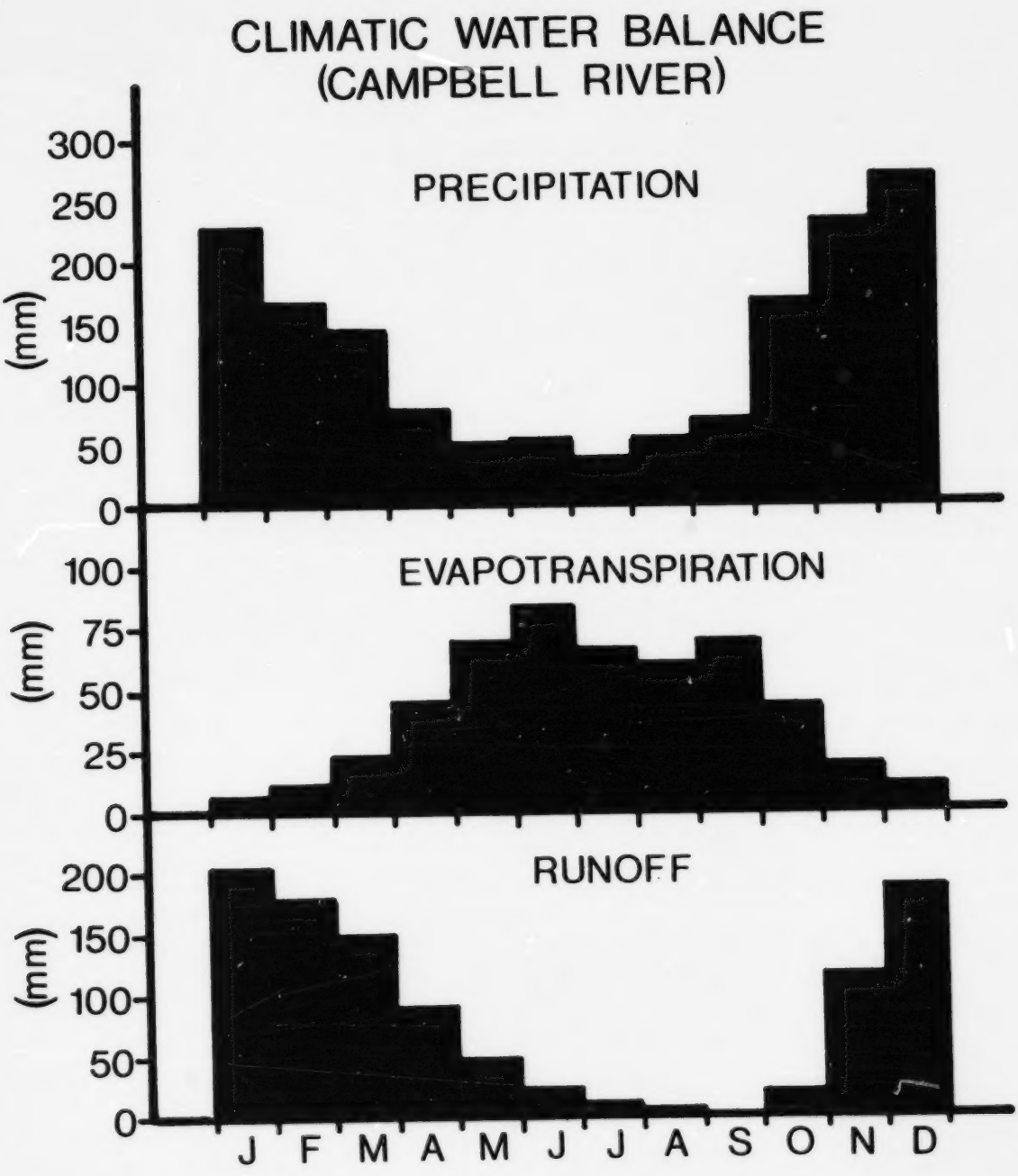

Fig. 4. Climatic water balance for study lakes (precipitation data from Campbell River airport).

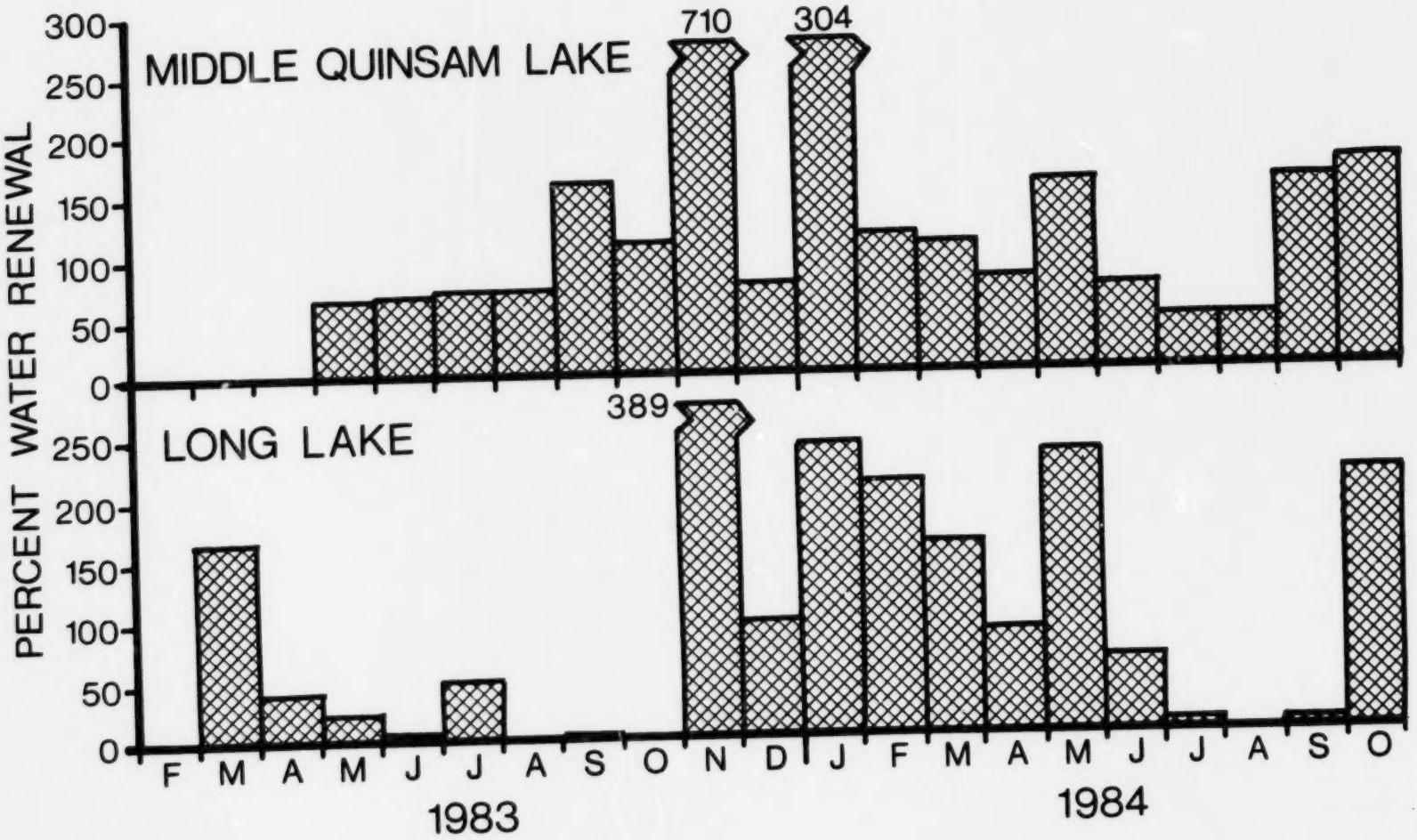

Fig. 5. Monthly water renewal rates (percent) for the study lakes.

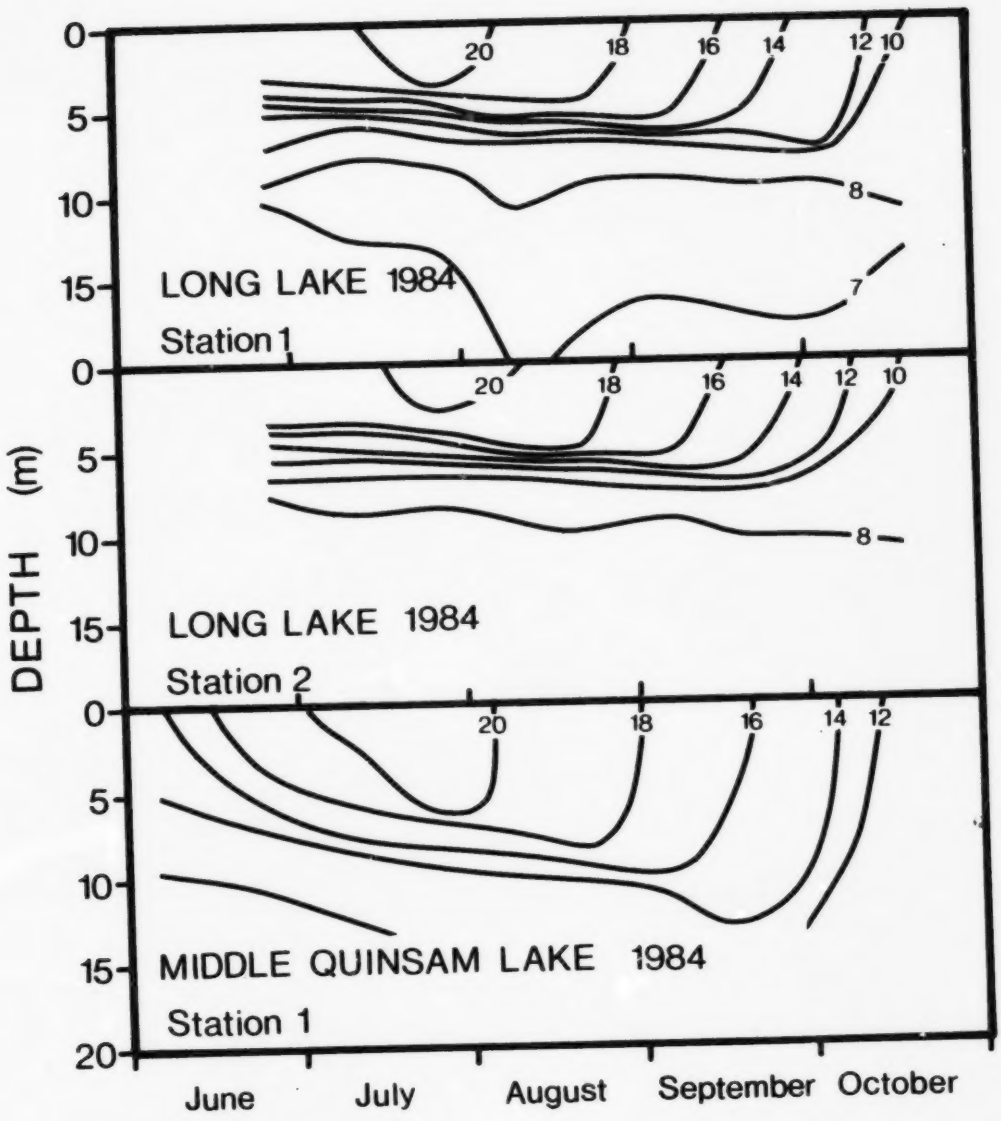

Fig. 6. Seasonal isotherms for 1984 from Long Lake (stations 1 and 2) and from Middle Quinsam Lake.

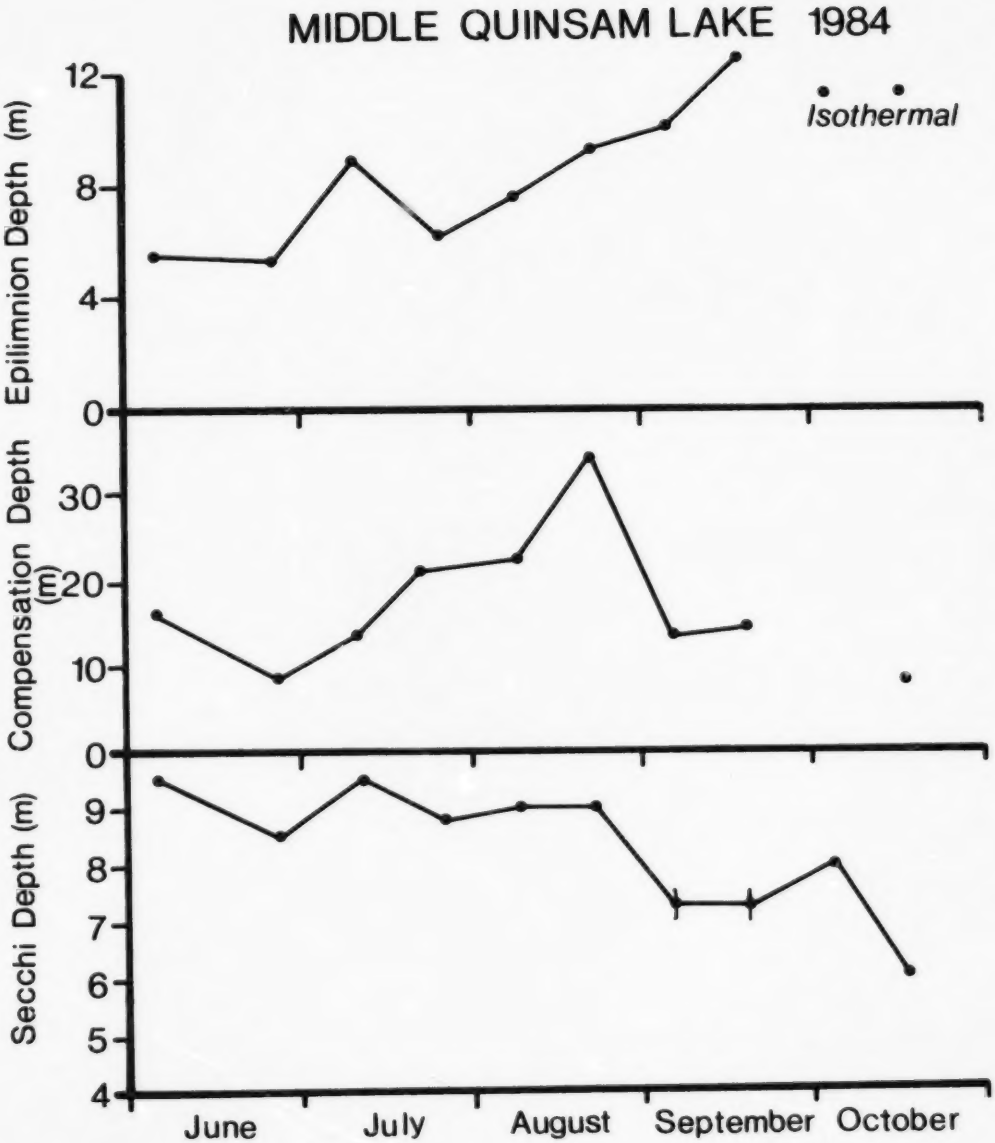

Fig. 7. Variation in water transparency and epilimnion depth during 1984 in Middle Quinsam Lake.

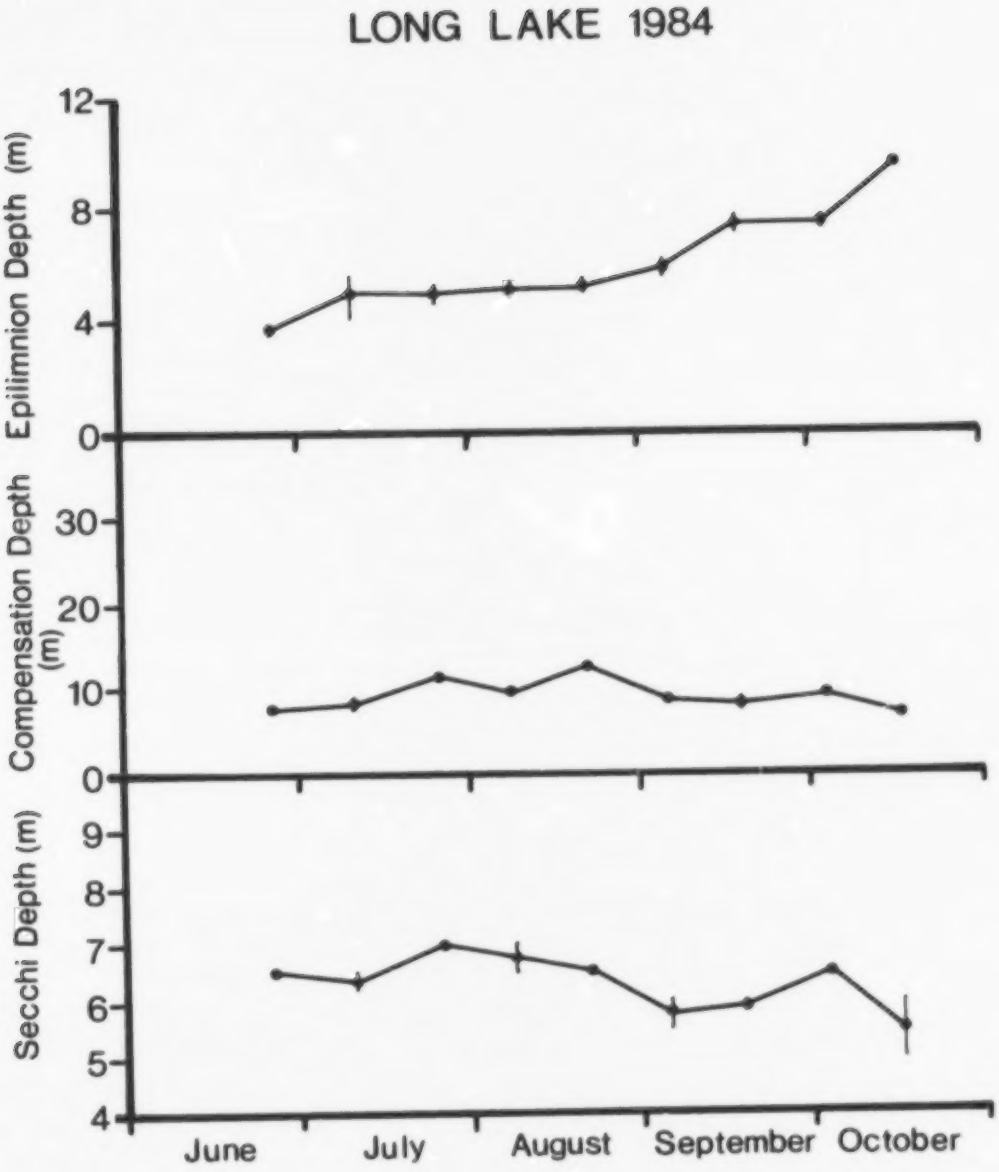

Fig. 8. Variation in water transparency and epilimnion depth during 1984 in Long Lake.

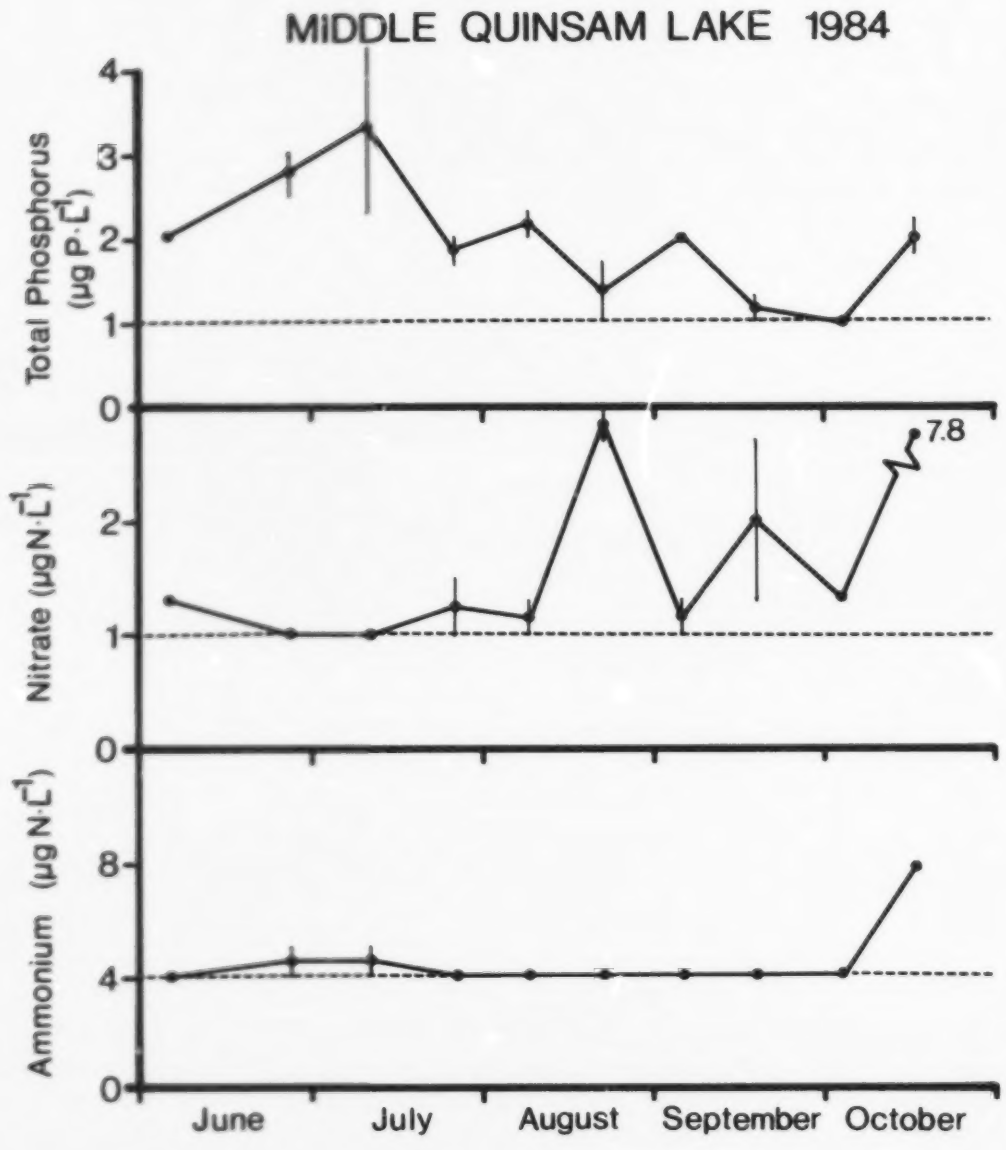

Fig. 9. Variation in total phosphorus, nitrate, and ammonium concentrations in Middle Quinsam Lake during 1984.

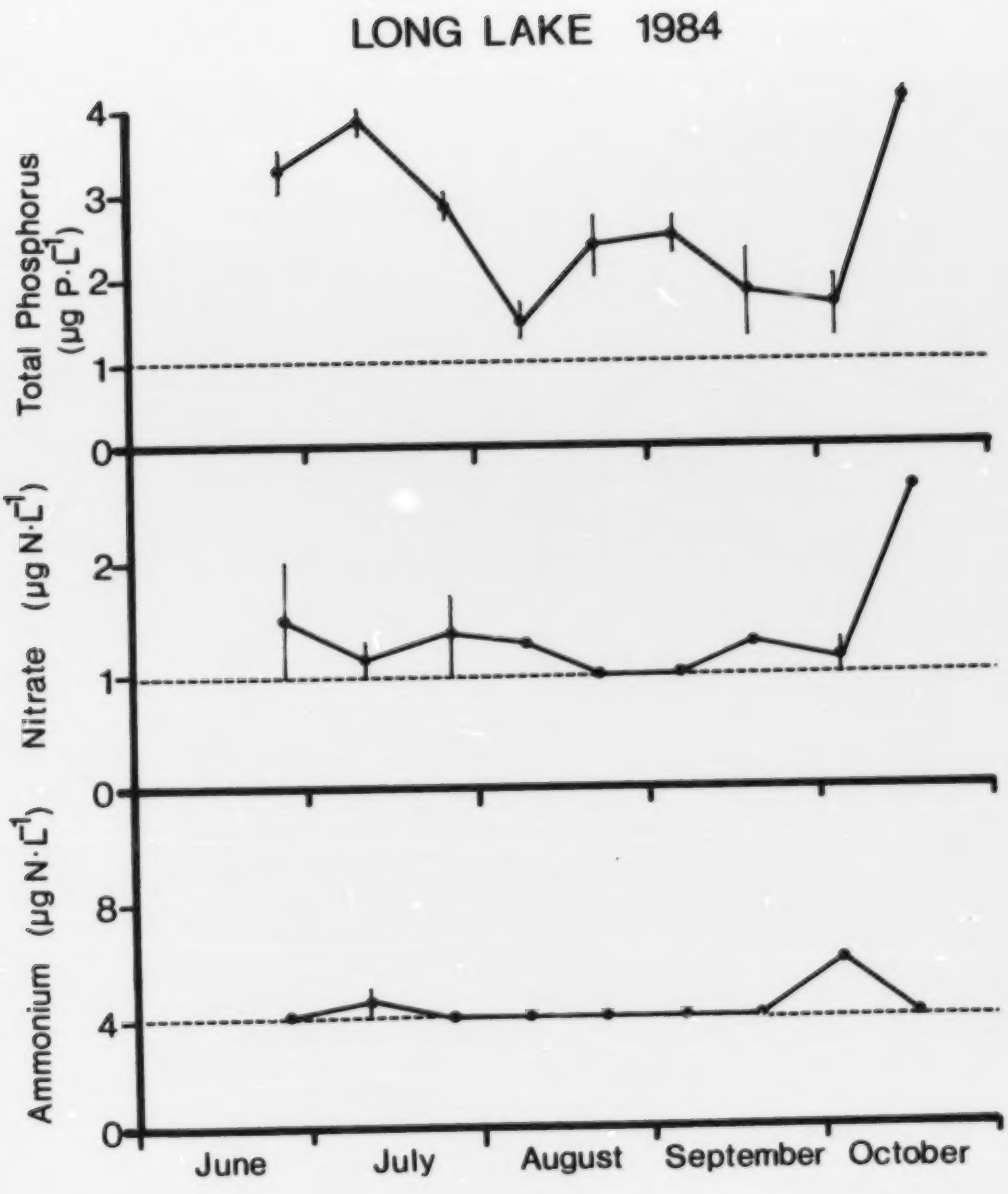

Fig. 10. Variation in total phosphorus, nitrate, and ammonium concentrations in Long Lake during 1984.

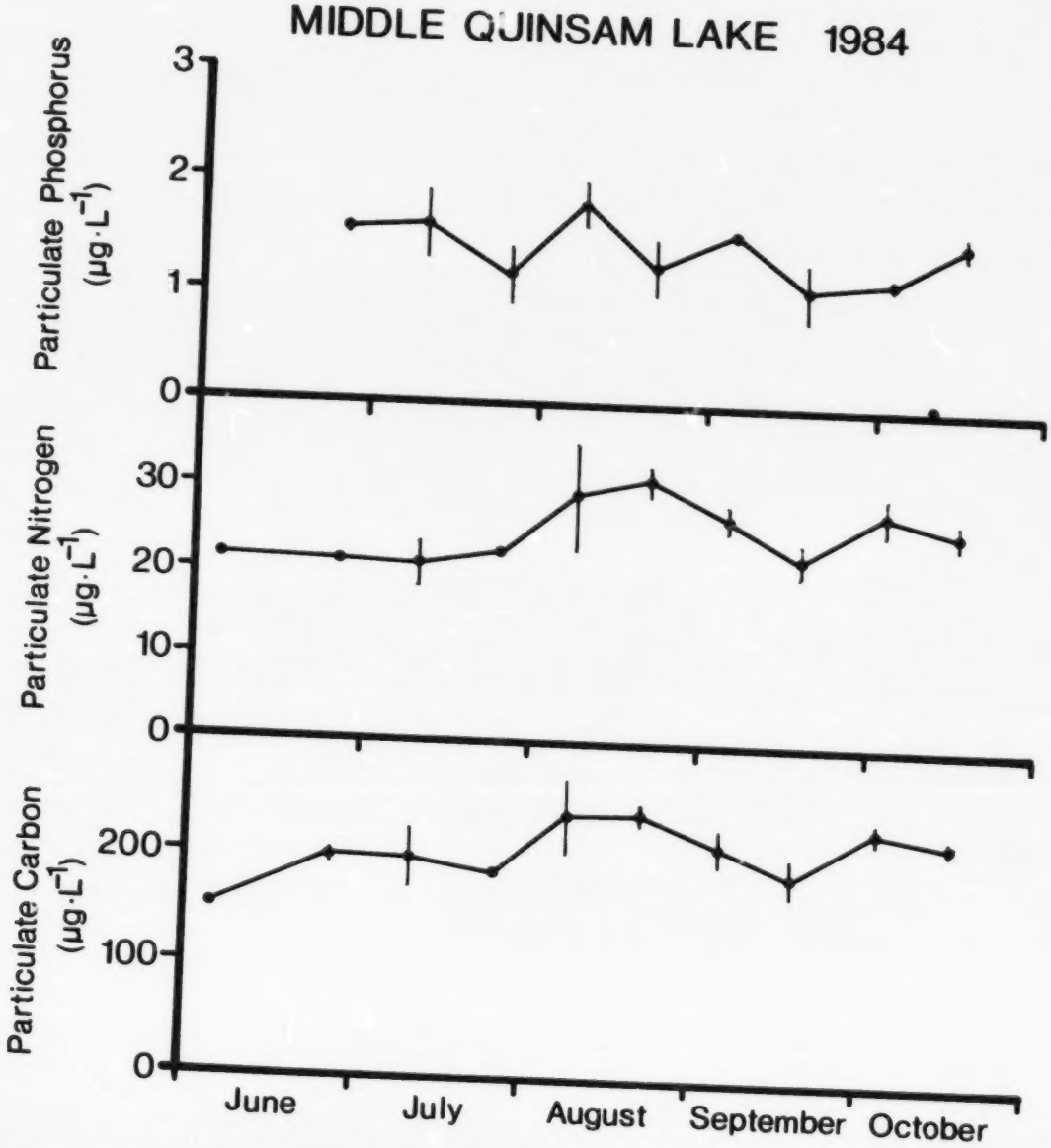

Fig. 11. Variation in particulate phosphorus, nitrogen, and carbon concentrations in Middle Quinsam Lake during 1984.

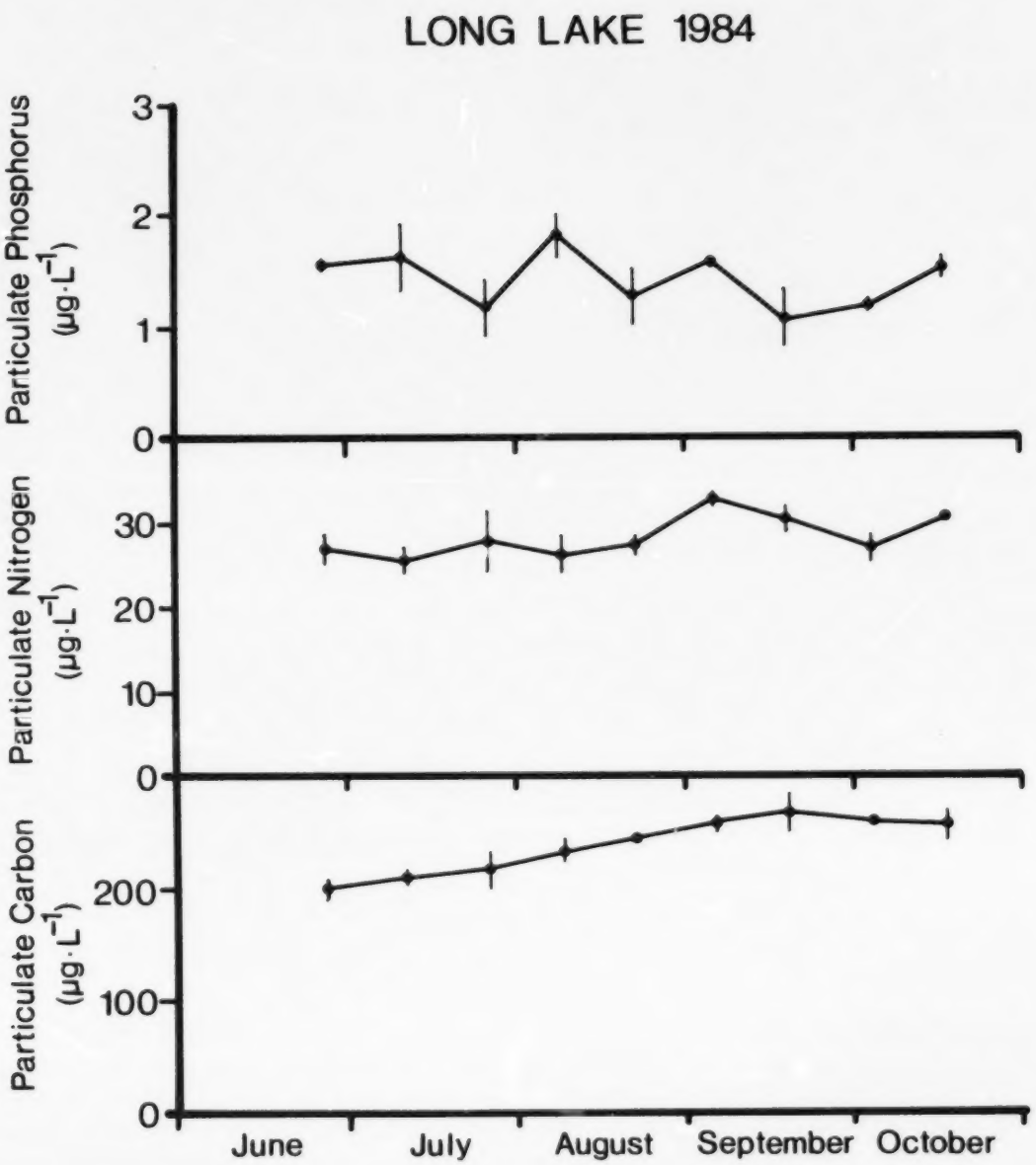

Fig. 12. Variation in particulate phosphorus, nitrogen, and carbon concentrations in Long Lake during 1984.

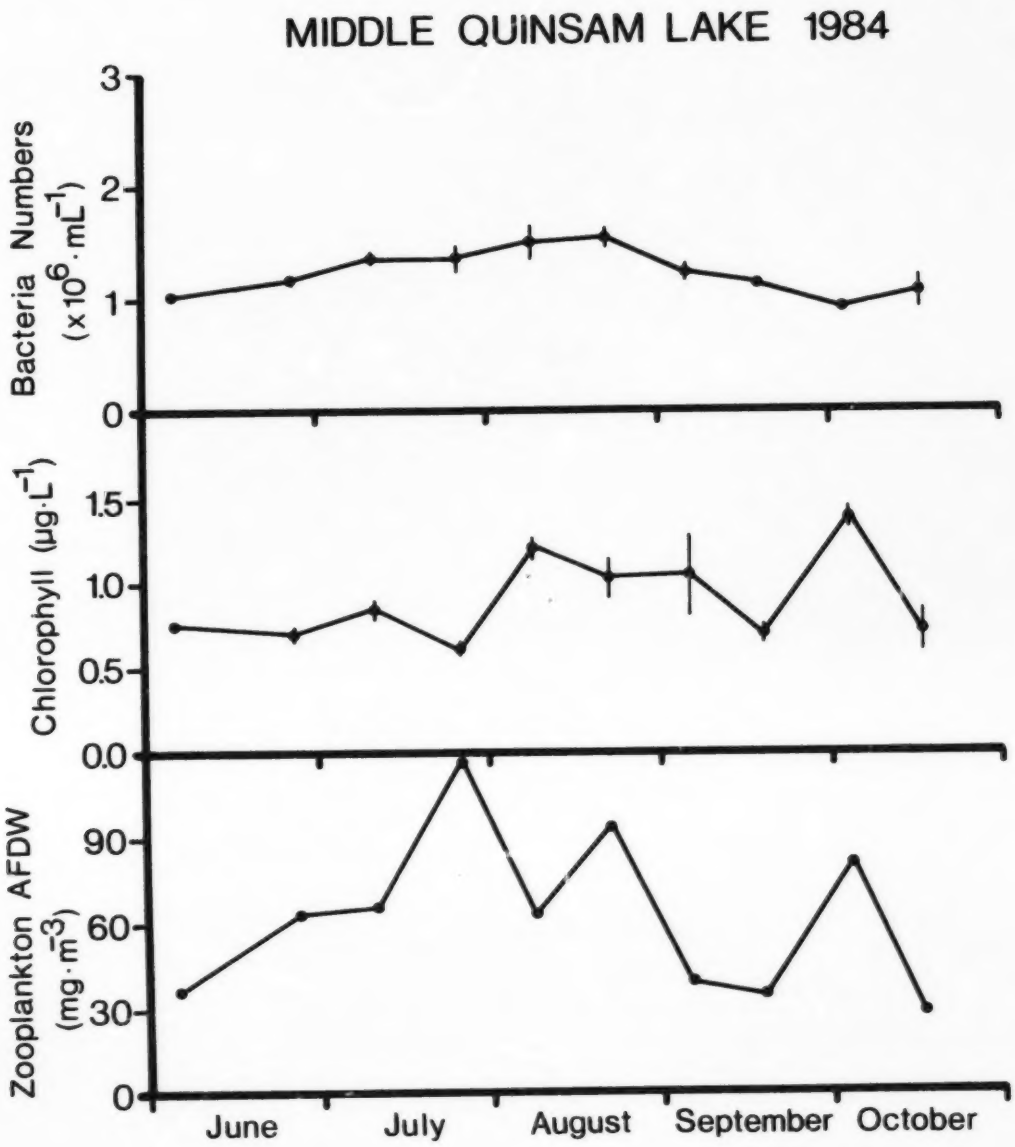

Fig. 13. Variation in bacteria numbers, chlorophyll concentration,, and zooplankton AFDW in Middle Quinsam Lake during 1984.

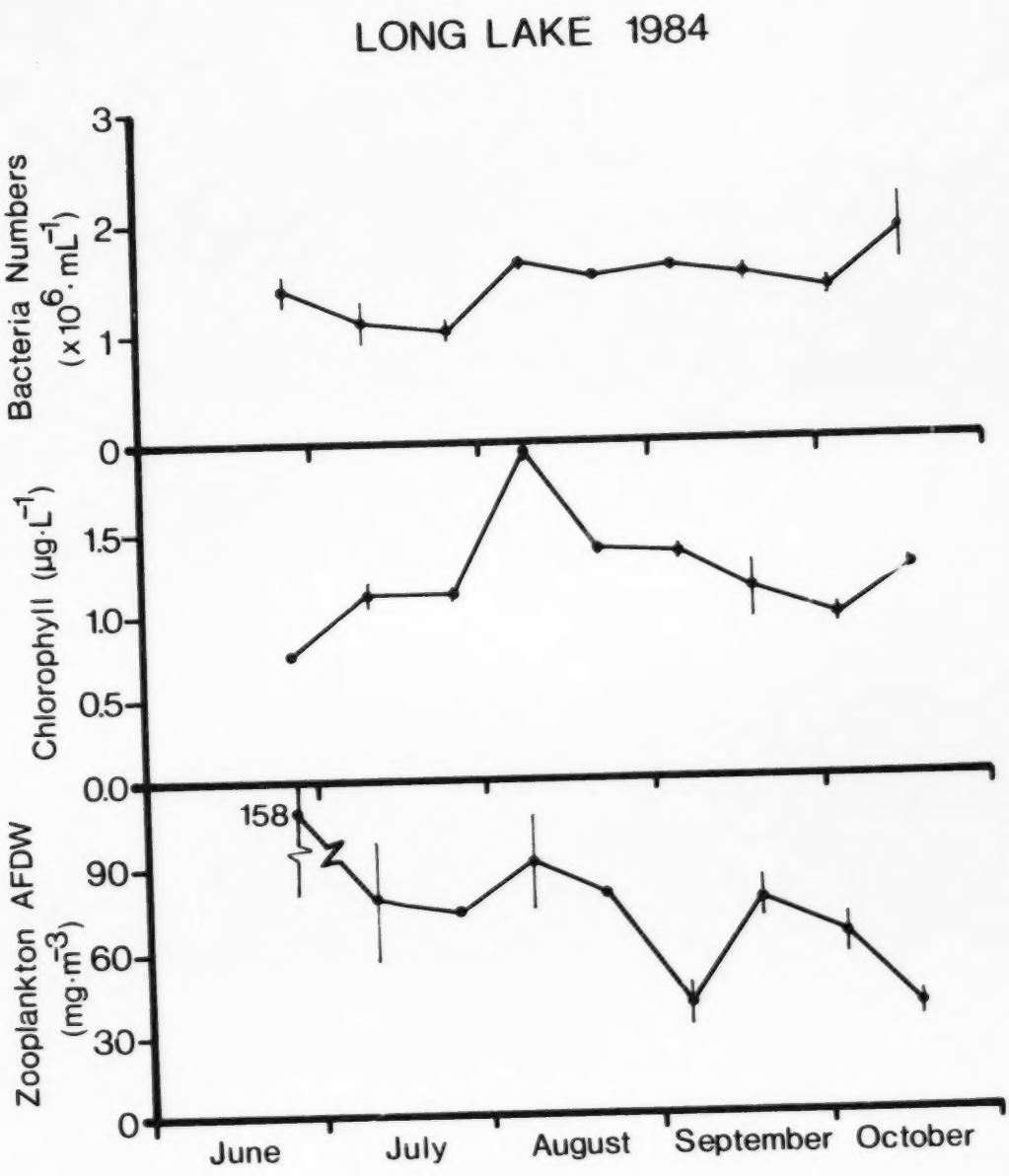

Fig. 14. Variation in bacteria numbers, chlorophyll concentration, and zooplankton AFDW in Long Lake during 1984.

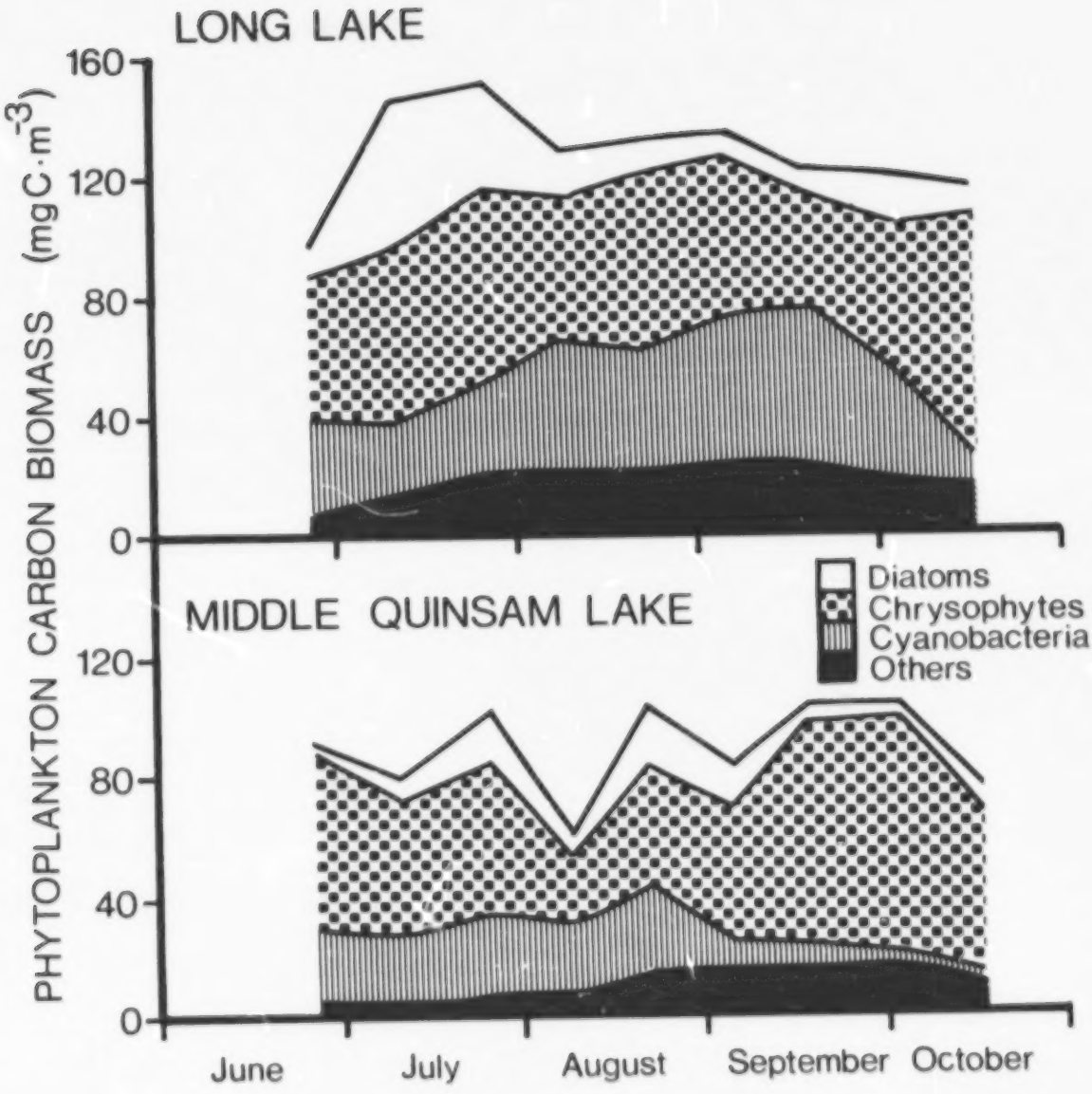

Fig. 15. Variation in phytoplankton carbon concentrations for four taxonomic groups in the study lakes during 1984.

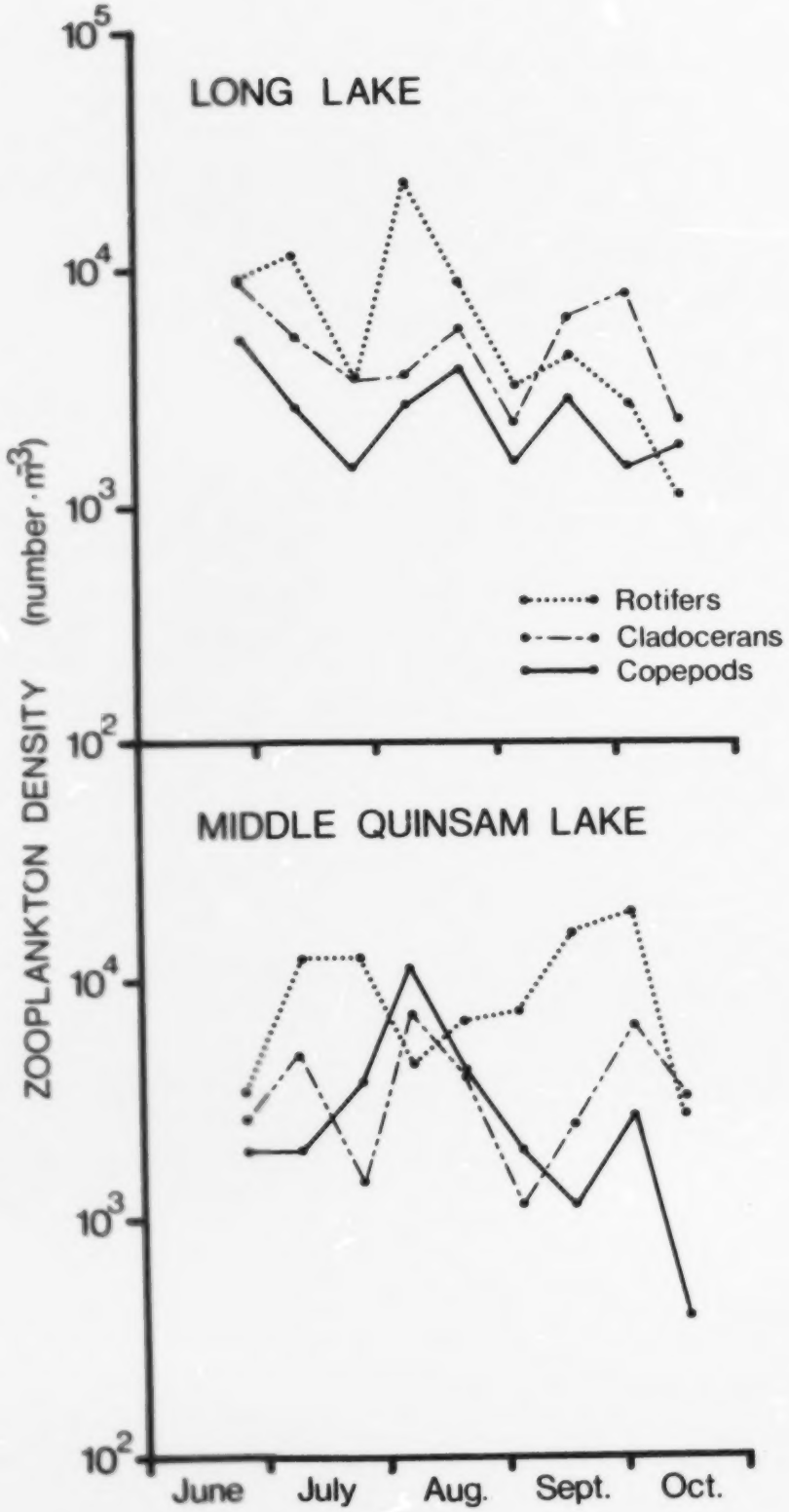

Fig. 16. Variation in densities of major zooplankton groups in the study lakes in 1984.

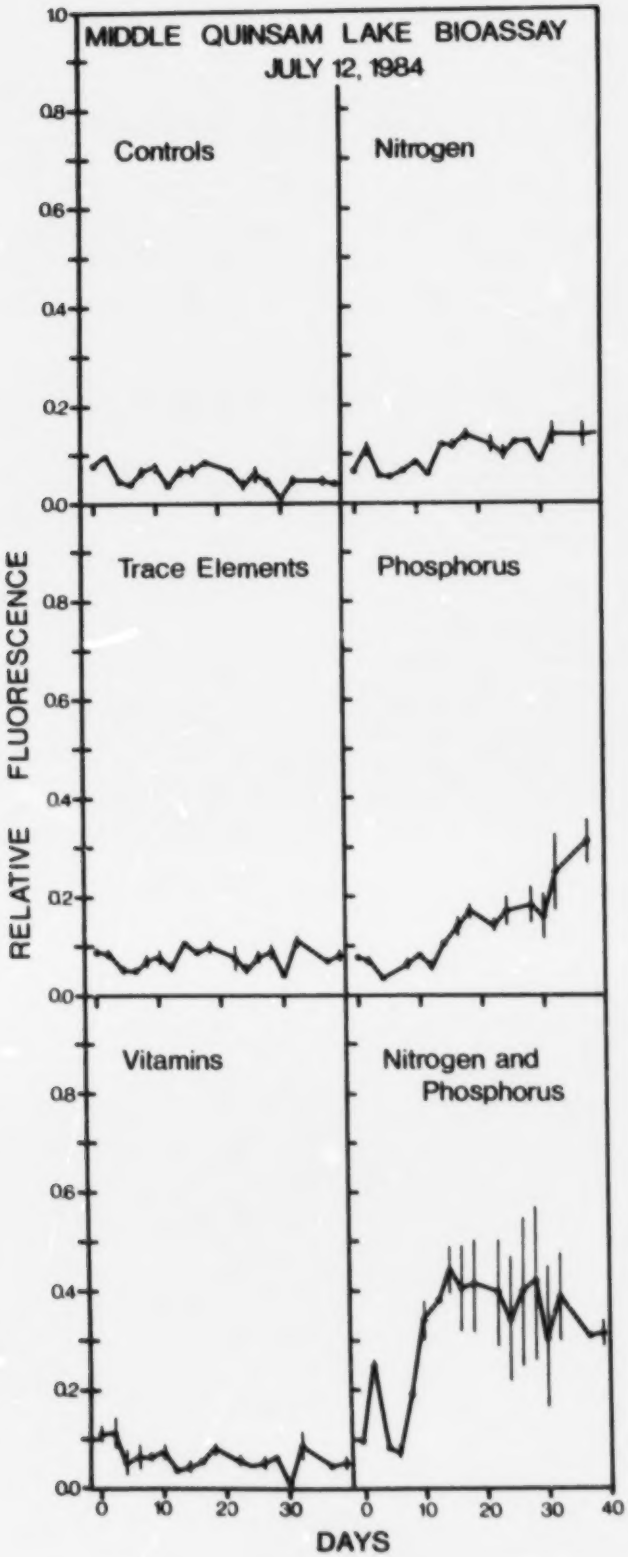

Fig. 17. Results from enrichment bioassays in Middle Quinsam Lake (water collected July 12, 1984).

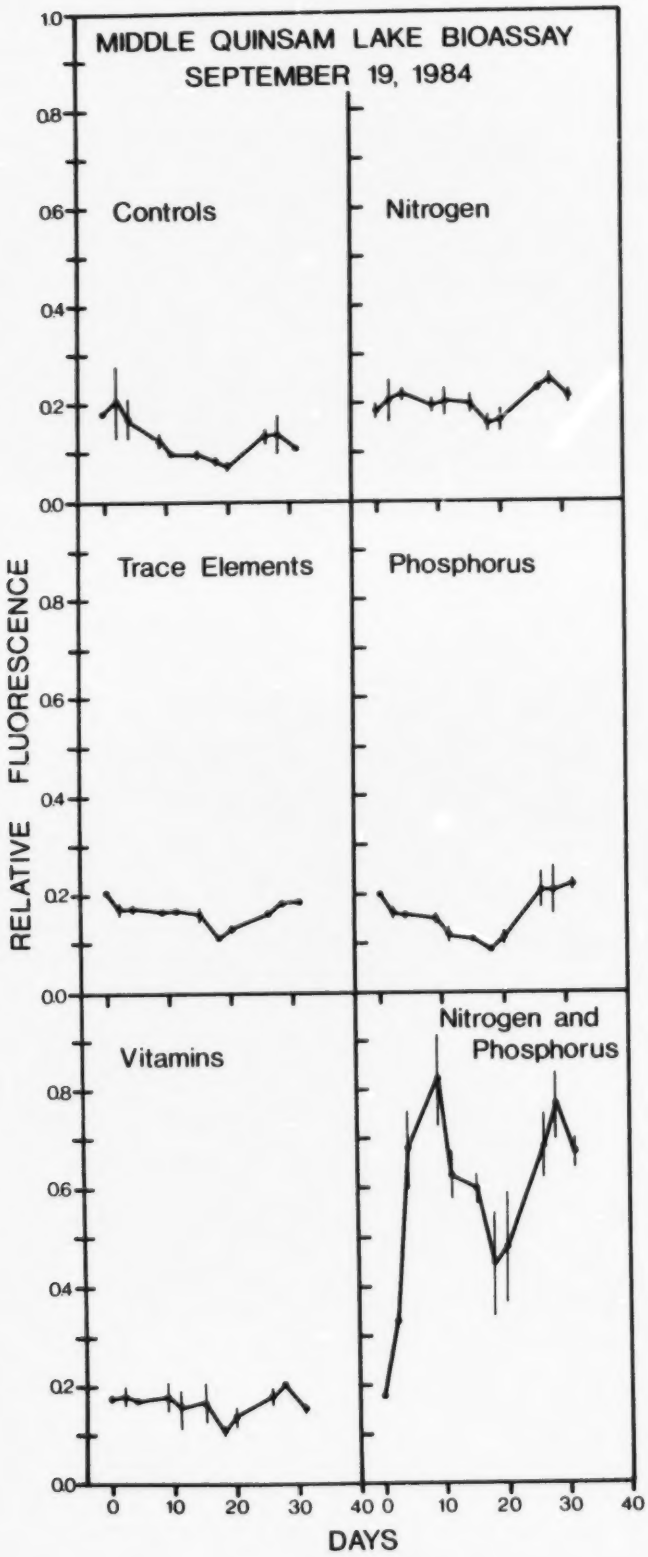

Fig. 18. Results from enrichment bioassays in Middle Quinsam Lake (water collected September 19, 1984).

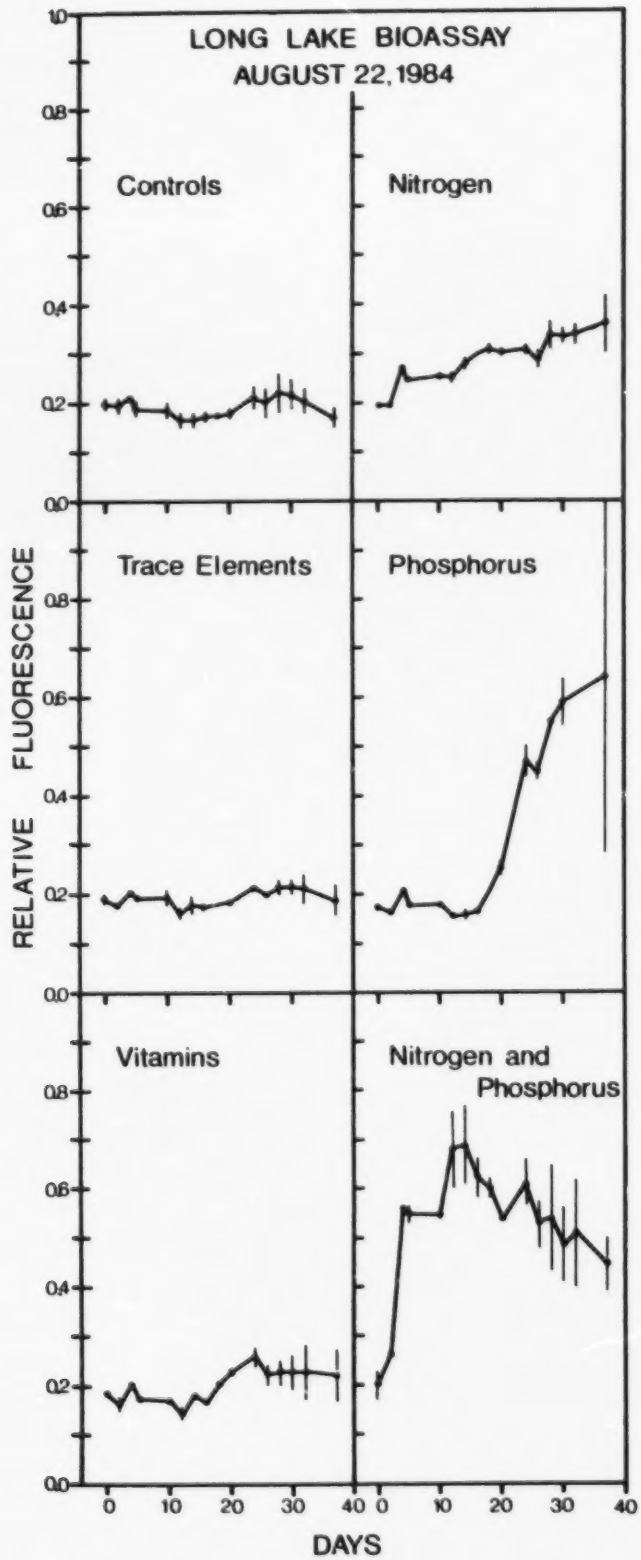

Fig. 19. Results from enrichment bioassays in Long Lake (water collected August 22, 1984).

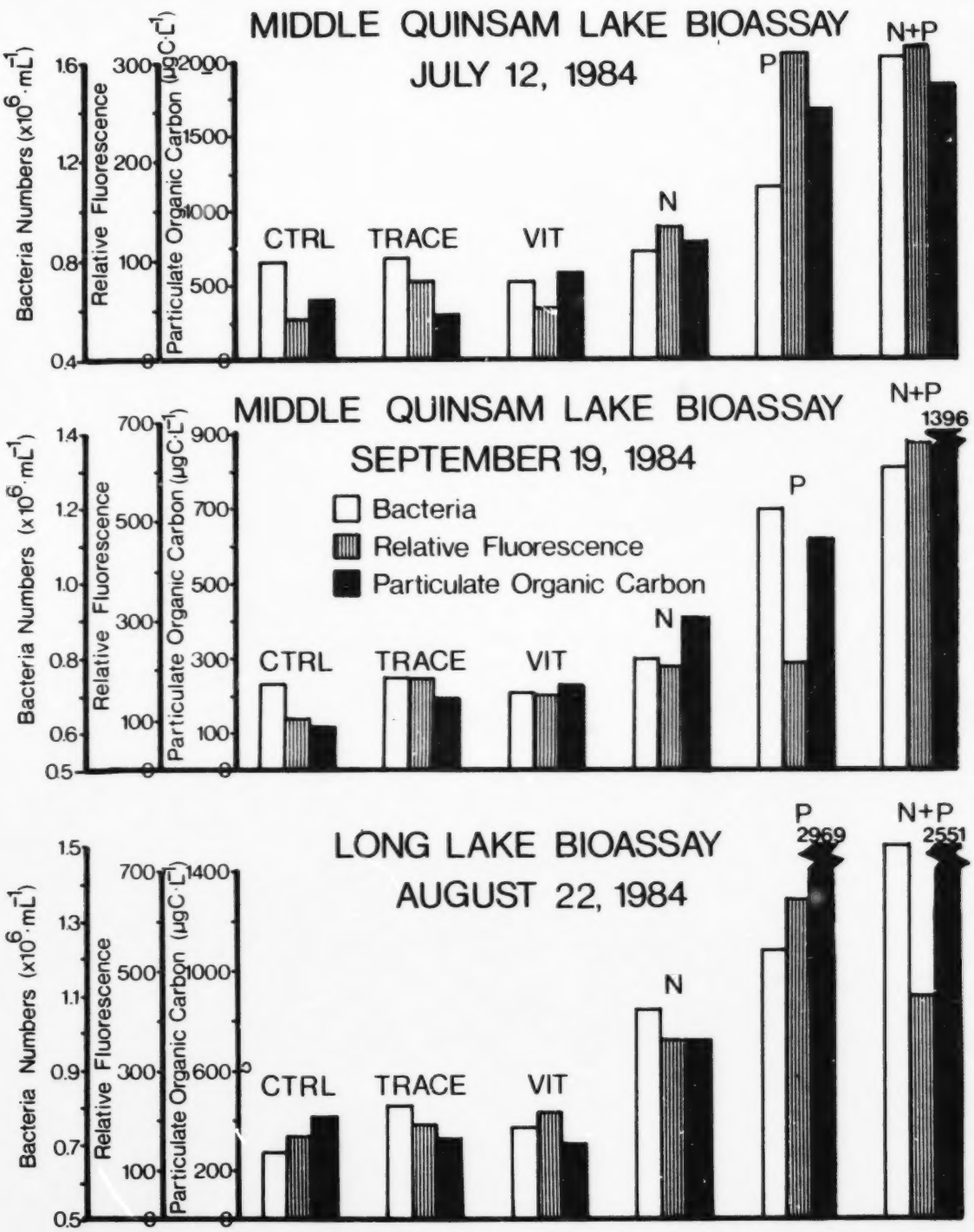

Fig. 20. Final yields from bioassay experiments (bacteria numbers, relative fluorescence, particulate organic carbon concentration).

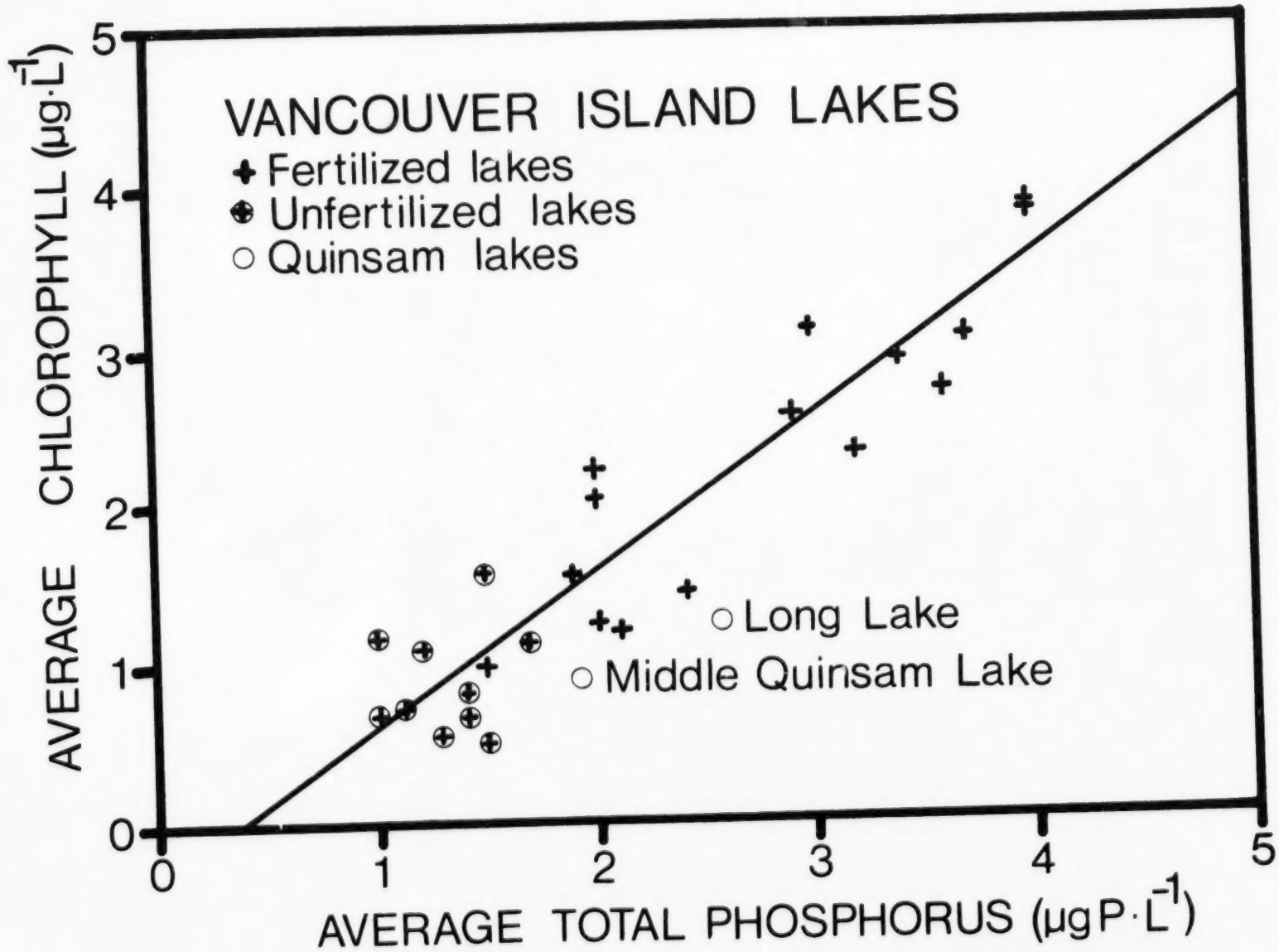

Fig. 21. Chlorophyll:phosphorus yields for Middle Quinsam and Long lakes compared to other oligotrophic and nutrient-enriched Vancouver Island lakes.

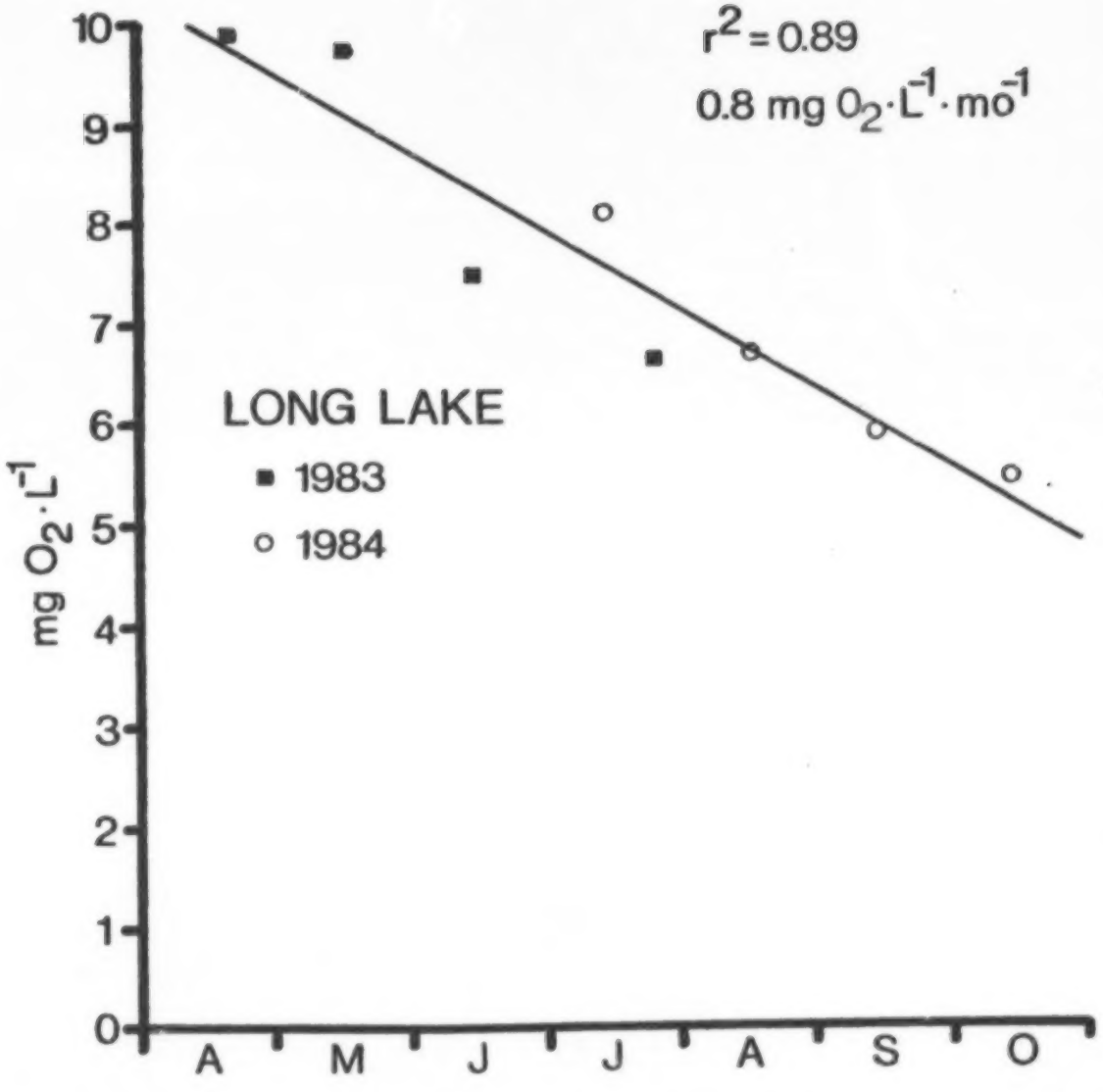

Fig. 22. Regression of hypolimnetic oxygen concentrations versus time in Long Lake in 1983 and 1984.